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Life-Cycle Impact Assessment:

A Conceptual Framework, Key Issues, and Summary of Existing Methods

Office of Air Quality
Clean Air
OAQPS
Planning and Standards

DISCLAIMER

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PREFACE

Life-cycle assessment (LCA) results can vary depending on how the sponsoring group defines the goals and scope of the LCA and what methods and data are used to conduct the assessment. There are an increasing number of organizations using LCA for a wide variety of internal and external purposes. Conducting an LCA can be complex, and may require significant data and information depending on the scope and goals of the study. For these reasons it appears desireable to develop scientifically based guidelines for conducting LCAs. Also, it is useful to provide technical reports to help users understand the status of LCA, available methods, sources of data, and other information relevant to conducting LCAs.

The U.S. Environmental Protection Agency (EPA) has responded by supporting a multioffice LCA program to develop technical information reports and, in some cases, various guidelines. This multioffice program consists of representatives from the Office of Research and Development (ORD), the Office of Air Quality Planning and Standards (OAQPS), the Office of Solid Waste (OSW), and the Office of Pollution Prevention and Toxics (OPPT). The LCA program uses a consensus-building approach, coordinating closely with the Society of Environmental Toxicology and Chemistry (SETAC). Through the organization of a series of workshops, SETAC has laid the groundwork for the development of a technical framework for conducting LCAs.

The first in a series of EPA LCA methodological guidelines documents, *Life Cycle Assessment: Inventory Guidelines and Principles*, and *Life Cycle Design Manual: Environmental Requirements and the Product System* were released in early 1993. Supplementary LCA documents including *Life-Cycle Assessment: Public Data Sources for the LCA Practitioner* and *Guidelines for Assessing the Quality of Life-Cycle Inventory Data* were released in April 1995. Ongoing EPA LCA projects include life-cycle inventory case studies on residential carpeting systems, shop towels in industrial laundries, and solvent alternatives; streamlined LCA methodology development; and product re-design through LCAs.

This document, which is a technical information report, includes the output from a two-phased research approach on the impact assessment component of LCA. Phase I identified and discussed key issues in the development of a conceptual framework for conducting an impact assessment. Phase II included documenting existing methods that exhibit potential for application in impact assessment and identifying gaps in the impact assessment methodology. This document contains the combined output of Phases I and II.

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CHAPTER 1

INTRODUCTION

Although a wide variety of impact assessment techniques have been integral to various disciplines, impact assessment in the context of Life-Cycle Assessment (LCA) is in its infancy. A conceptual framework for conducting impact assessment has been established, but experts have not yet reached a consensus on specific methods and procedures. This document outlines a possible framework, discusses key issues, and summarizes existing methods for conducting impact assessment. This document is not a guidance document, however, but rather a compendium on the state of practice of impact assessment.

LCA is a holistic concept and methodology for evaluating the environmental and human health burdens associated with a product, process, or activity. A complete LCA identifies inputs and outputs; assesses the potential impacts of those inputs and outputs on ecosystems, human health, and natural resources; and identifies opportunities for achieving improvements. The basic life-cycle stages covered in LCA include raw materials acquisition, manufacturing, use/reuse/maintenance, and recycling/waste management. The LCA approach consists of four interrelated components, including impact assessment. These components are illustrated in Figure 1-1 and explained below.

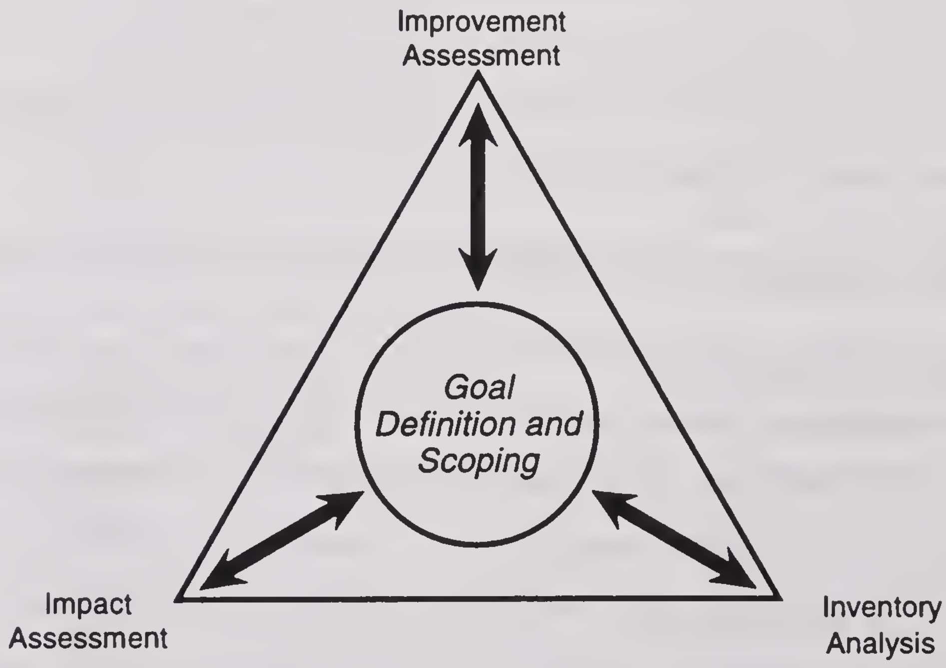


Figure 1-1. LCA Conceptual Framework

- **Goal definition and scoping:** the explanation of the study's purpose and objectives; the identification of the product, process, or activity of interest; the identification of the intended end-use study results; and the key assumptions and methods employed.
- **Inventory analysis:** the identification and quantification of raw materials and energy inputs, air emissions, water effluents, solid waste, and other life-cycle inputs and outputs.
- **Impact assessment:** the qualitative or quantitative classification, characterization, and valuation of impacts of the inventory items to ecosystems, human health, and natural resources, based on the results of an inventory analysis and application of various methods and models to determine significance of the inventory items.
- **Improvement assessment:** the identification and evaluation of opportunities to achieve improvements in products and/or processes that result in reduced environmental impacts, based on the results of an inventory analysis or impact assessment.

For almost 20 years, a wide variety of organizations have conducted less-than-complete LCAs. Most of these LCAs focused on the inventory analysis component and stopped short of analyzing impacts. This focus has enabled LCA analysts to concentrate on developing and refining procedures for building credible and reliable inventories of system inputs and outputs and using these inventories for identifying possible improvement opportunities.

Formal procedures for conducting impact assessments have not yet been established. The primary purpose of impact assessment in LCA is to assess the potential impacts resulting from inputs and outputs quantified in the inventory analysis. By providing this information, impact assessment can enhance the basis for evaluating and justifying the trade-offs among a variety of inputs and outputs, as well as improvement options. As existing LCA and impact assessment tools are refined and new ones developed, practitioners are expected to include more impact assessments as part of LCAs.

1.1 KEY IMPACT ASSESSMENT TERMS AND CONCEPTS

In developing procedures for impact assessments, an important step is establishing a common language. Fundamental terms used in impact assessment are often the subject of confusion. For example, distinguishing between an inventory item and an impact is not always easy. Although a common practice is to account for the amount of solid waste materials produced by a system in an inventory analysis, it is not common to account for the amount of natural habitat consumed to dispose of that solid waste. Some analysts might consider this consumption of natural habitat an input, while others might consider it an impact. This section focuses on key terms that distinguish between inventory item and impact and provides a working

definition of impact assessment. Table 1-1 defines an inventory item and an impact and lists examples and issues related to each term. Appendix C provides a glossary for other terms and concepts used in this document.

1.1.1 Inventory Item

An inventory item is defined in this document as a quantitative measure of an energy or raw material requirement, atmospheric emission, waterborne effluent, solid waste, or other input or output of a particular product, process, or activity. In the past, an inventory item has referred to more traditional inputs and outputs. For purposes of impact assessment, however, some more nontraditional inputs and outputs (e.g., soil compaction, habitat use) associated with a production system also may be appropriate to consider in the inventory analysis.

TABLE 1-1. KEY IMPACT ASSESSMENT TERMS: DEFINITIONS, EXAMPLES, AND ISSUES

	Inventory Item	Impact
Definition	A quantitative measure of an energy or raw material requirement, atmospheric emission, waterborne effluent, solid waste, or other quantifiable input or output of a particular product, process, or activity.	An actual or potential change in an environmental characteristic resulting from interactions between the inventory items, or components of a particular product, process, or activity and the environment.
Examples	• tons of SO_2 emissions/year • tons of solid waste per day • biochemical oxygen demand (BOD) released per unit of production • tons of oil per unit of output	• acid precipitation • ozone depletion • soil erosion • habitat consumption • increased risk of cancer
Issues Related to Impact Assessment	• Interaction between different releases may create new substances that increase or mitigate effects. • Uncertainty of inventory data can dramatically affect the results of impact assessment.	• Uncertainty is associated with the existence, nature, and extent of impacts in an uncontrolled environmental setting. • Multiple impact pathways make allocating impacts difficult. • Qualitative items, such as habitat consumption and social welfare, are difficult to determine and quantify.

One issue associated with inventory items in the context of LCA is that the composition of the inventory is primarily based on the goals and scope of the study. Because every input and output of a production system cannot typically be included in the inventory analysis, those included and/or excluded from the scope of the inventory analysis should be made transparent to the user. In addition, practitioners may want to modify the goals and scope of the study to see how that modification affects not only the composition of inputs and outputs captured in the inventory analysis, but the overall LCA as well.

A second issue associated with inventory items is the synergistic nature of some compounds. The synergistic effect of mixed compounds may increase the concern about the original compound or create a new compound(s) that is not captured in the inventory. Such synergistic compounds may have the potential to create combined impacts greater than those of the individual releases. For example, the interaction between sulfur dioxide (SO_2) and particulate matter—where small particles transport SO_2 and sometimes sulfuric acid deep into the lungs—can increase damage (Ott, 1987). Synergistic compounds do not necessarily need to be included in the inventory, but practitioners should nonetheless recognize this potential effect and other factors (e.g., antagonistic effects, assimilative capacity) when drawing conclusions based on LCA results.

Another issue is distinguishing between an inventory item and an impact. For instance, should a largely qualitative item such as habitat consumption be included as an inventory item or should it be treated as an impact? For purposes of this document inventory items are limited to readily quantifiable "traditional" inputs and outputs (e.g., raw materials and energy use, air emissions, waterborne effluent, solid waste). Items such as habitat consumption that are not so easily quantified and often involve value judgments are treated as impacts.

1.1.2 Impact

In the context of LCA, impact may be defined as an actual *or* potential change in an environmental characteristic that results from interactions between the components of a defined system and the environment. Impacts relevant to impact assessment are categorized according to whether they affect ecosystems, human health, and natural resources (SETAC, 1993). Although they are not the primary focus of impact assessment, social welfare impacts may also be considered to the extent that they indirectly may cause impacts to ecosystems, human health, and natural resources. Currently, methods for handling social welfare impacts in the context of impact assessment are not well developed.

The Stressor Concept

Although not explicitly used in this document, the stressor concept has provided a useful means of talking about the relationship between inventory items and subsequent impacts. A stressor is defined as any physical, chemical, or biological entity that can induce an impact, and may be characterized by the following attributes:

- Type: chemical, physical, or biological
- Intensity: concentration, magnitude, abundance/density
- Duration: acute (short term) versus chronic (long term)
- Frequency: single event versus recurring or multiple exposures
- Timing: time of occurrence relative to environmental and human health parameters
- Scale: spatial extent and heterogeneity in intensity (EPA, 1992c).

The stressor concept is imbedded (implicitly) in life-cycle impact assessment. In this context, a stressor can be an inventory item that leads to a primary impact(s), or a stressor can be an impact that leads to a secondary impact(s), and so on. For example, a stressor could be identified as a quantity of SO_2 emissions to the air from a given product or process system. This SO_2 can be linked to primary impacts such as acid precipitation. Acid precipitation is an impact of SO_2 emissions as well as a stressor, because it can be linked to secondary impacts such as acidification of water bodies, tree damage, building materials corrosion, and the leaching of metals from soils.

Several issues are related to the definition of an impact. First, impact in the context of impact assessment rarely means an actual impact but instead means a *potential* impact. The term "potential" is not meant to minimize concern for those impacts but to point out that impact assessment does not necessarily provide direct measures of actual impacts, such as the actual number of dead fish that result from the waterborne effluent X of process A. Instead, impact assessment might attempt to establish a link between inventory items and potential impacts. For example, waterborne effluent X from process A may be identified in the literature as toxic to fish above a threshold concentration. Researchers can use this threshold to indicate the potential for impact and not the actual number of fish harmed or killed. Thus, unless otherwise specified, the term "impact" in this report implicitly carries the connotation of *potential* impact.

A second issue is the difficulty in quantifying potential impacts (e.g., estimating the number of fish mortality resulting from release of waterborne effluent X). Limitations in data

availability, modeling, and resource limitations—and the complexity of most natural systems—often require a more qualitative description of impacts based on some amount of quantitative information (e.g., level of pollutants released). This issue, however, should not discourage practitioners from conducting impact assessments. Depending on the goals and scope of the LCA, qualitative information may be adequate, and in some cases preferred, to assist users in identifying and evaluating opportunities to achieve environmental improvements.

A third issue associated with the term impact is the potential large number of impacts associated with any given inventory item. That is, although impact assessment attempts to establish a link between inventory items and impacts, a large number of impacts can be associated with any single inventory item. Ideally, impact assessment would analyze every potential impact, but that would typically be infeasible. Therefore, practitioners need to decide which impacts are within the goals and scope of the LCA and if those impacts can be estimated or measured.

Finally, the potential for an impact to occur is not easily defined, nor easily captured, in any impact analysis. The analysis is hindered by a number of uncertainties and a general lack of knowledge about the natural processes that determine the fate, or impact, of substances or activities in the environment. The potential for an impact to occur is governed by a number of different variables, such as those listed in the following function:

Impact = f (location, medium, time, rate of release, routes of exposure, natural processes, persistence, mobility, accumulation, toxicity, concentration of release, assimilative capacity, synergism, antagonism, etc.)

The uncertainty associated with an impact actually occurring is often the subject of considerable debate. Uncertainty, in this context, focuses on the interrelationships between the inventory items and the associated impacts and between the impacts themselves.

1.1.3 Impact Assessment

In the LCA literature, impact assessment has different meanings for different people. The following are a few examples of the multiple interpretations of impact assessment presented in the context of LCA:

- An assessment of the impacts on human health and the environment associated with raw materials and energy inputs and environmental releases quantified by the inventory (Tellus Institute, 1992a).

- A system utilized to ascertain the elements and processes involved in translating impact indicators into the response of environmental receptors and the associated impacts incurred by the receptors and suffered within the process of transformation (Canadian Standards Association, 1992).
- A technical, quantitative, and/or qualitative process that characterizes and assesses the effects of environmental loadings as identified in the inventory stage of the LCA (SETAC, 1993).
- A process that meaningfully relates inventory information into relevant concerns about natural resource usage and potential effects of environmental loadings, consistent with the defined scope, specificity, and technical precision of the life-cycle inventory data (Procter and Gamble, 1992).
- An analysis of the effects of inputs and outputs on the environment, where the effects are secondary inventory values that are induced to change as a result of the primary inputs and outputs of an industrial system (Scientific Certification System, 1992).

An underlying theme throughout these descriptions is that impact assessment is a process of linking the inputs from and outputs to the environment (which are compiled in the inventory) to potential impacts to ecosystems, human health, natural resources, and possibly social welfare impacts. For purposes of this report we define impact assessment as follows:

Impact assessment: A systematic process to identify, characterize, and value potential impacts to ecosystems, human health, and natural resources based on the results of a life-cycle inventory.

1.2 PURPOSE OF LIFE-CYCLE IMPACT ASSESSMENT

Impact assessment attempts to take the input and output data compiled in an inventory analysis and translate that data into either (1) a quantitative and/or qualitative description of environmental impact, or (2) a description of how each inventory item (per functional unit) contributes to environmental impacts. A complete impact assessment considers potential impacts to the full range of environmental media (e.g., air, water, land).

Conceivably, LCA could stop after the inventory analysis. One reason it does not is that impact assessment makes explicit the methods used to compare and weigh inventory items. Failing to communicate these methods might convey that all inventory items have relatively similar magnitudes of impacts. Another reason for continuing past the inventory analysis is to provide the LCA user with information that is more useful for decisionmaking. For example, determining the relative overall environmental burden associated with two product systems is

often difficult when the emissions of one pollutant, say SO_2, are estimated to be higher for one production system, while emissions of a different pollutant, say reactive hydrocarbons, are estimated to be higher for the other production system.

LCA is not necessarily a linear or stepwise process. Rather, as suggested by Figure 1-1, information from any of the three components can complement information from the other two components. For instance, opportunities for environmental and human health improvements do not necessarily stem from the improvement assessment but can be realized at any stage of the LCA process. The inventory component alone may be used to identify opportunities for reducing the amounts of specific inputs and outputs. The impact assessment can provide additional information about the significance of the inventory items, or identify priorities for the improvement assessment. The impact assessment may also present important information suggesting modification of the goals and scope of the LCA, or identify data gaps, research needs, or significant uncertainties in the LCA. The following sections discuss the relationships between impact assessment and the inventory analysis and improvement assessment components of LCA.

1.2.1 Relationship Between Impact Assessment and Inventory Analysis

Impact assessment focuses on describing potential impacts to ecosystems, human health, and natural resources through the use of a variety of models. Typically, these models require supporting data (e.g., environmental or human health information). Therefore, the type of data collected in the inventory analysis must be commensurate with the impact assessment model(s).

Upgrading inventory data may be necessary to account for the specific data needs of an impact assessment. While conducting the impact assessment, a practitioner may realize that additional data (e.g., toxicological, environmental parameters) are needed. For example, to conduct a detailed impact assessment, the practitioner may need to have information on pollution speciation or geographic and temporal specificity of impacts. On the other hand, certain inventory data may not be required given the scope of the overall LCA and/or the impact assessment.

The importance of making goal statements and determining scope and boundary conditions prior to developing the inventory is critical. These activities ensure that the inventory has the appropriate data needed for conducting the impact assessment or that additional data collection has been planned, if needed. Inadequate planning can lead to needing additional data later, which may cause unplanned expenditures or the exclusion of items from the impact assessment.

1.2.2 Relationship Between Impact Assessment and Improvement Assessment

The purpose of the improvement assessment component of LCA is to identify and evaluate opportunities for reducing or mediating environmental impacts. Opportunities to achieve improvements may be identified at any stage of the LCA process. Impact assessment provides a means of identifying improvement opportunities on the basis of impacts. Although inventory results can be used to identify opportunities for improvement, impact assessment can take this information one step further to assess the impacts of the inventory. Also, an impact assessment supplements the improvement assessment by providing baseline information and identifying variables that will require further monitoring. Thus, the complexity of the impact assessment must be matched with the final end use of the resulting information from the improvement assessment. Once again, scoping plays a large role in maintaining consistency between the LCA components.

Options identified in the improvement assessment should be evaluated to ensure that the improvement programs do not create additional, unanticipated impacts. For example, during the improvement stage the practitioner may discover impacts from proposed improvements themselves that were not considered in the initial impact assessment. At that point, broadening the scope of the impact assessment may be necessary to account for the additional impacts.

Although adjusting the scope of the overall LCA or of each LCA component to meet unforeseen events is possible, maintaining a consistent scope across the components is desirable. This consistency ensures that the study uses resources and time efficiently and produces results that are consistent with the goals and objectives of the overall LCA.

1.3 APPLICATIONS OF IMPACT ASSESSMENT

In the context of LCA, impact assessment may be perceived as one tool that decisionmakers use in the LCA decision development and improvement implementation stages. As standard procedures and techniques for impact assessment are developed and refined, impact assessment will enhance the quality of the decision and provide the decisionmaker with a better frame of reference within which to make the decision.

In the present-day context of LCA, impact assessment may be useful for

- characterizing the environmental impacts of inventory items,
- uncovering significant cross-media transfers of impacts,
- incorporating environmental and human health concerns into the decisionmaking process,

- evaluating impacts for their relevance to predetermined LCA goals and objectives, and
- translating all impacts and their determined importance to the LCA audience in a clear and concise manner (Canadian Standards Association, 1992).

Specific applications of impact assessment extend beyond those of inventory analysis. Although an inventory analysis provides a quantified listing of inputs and outputs, an impact assessment relates these items to resulting environmental impacts in a meaningful manner. For purposes of this document, two general types of LCA applications are distinguished:

1. **Internal applications** — where results are used within an organization and are not intended for public release; and
2. **External applications** — where results are used, or are intended for use, in a more public context.

As shown in Figure 1-2, the scope and degree of quantification generally increase in moving from internal to external LCA applications. The broader scope and higher degree of quantification is often needed for externally applied studies that must withstand widespread public scrutiny. Table 1-2 provides an overview of a range of internal and external applications of impact assessment.

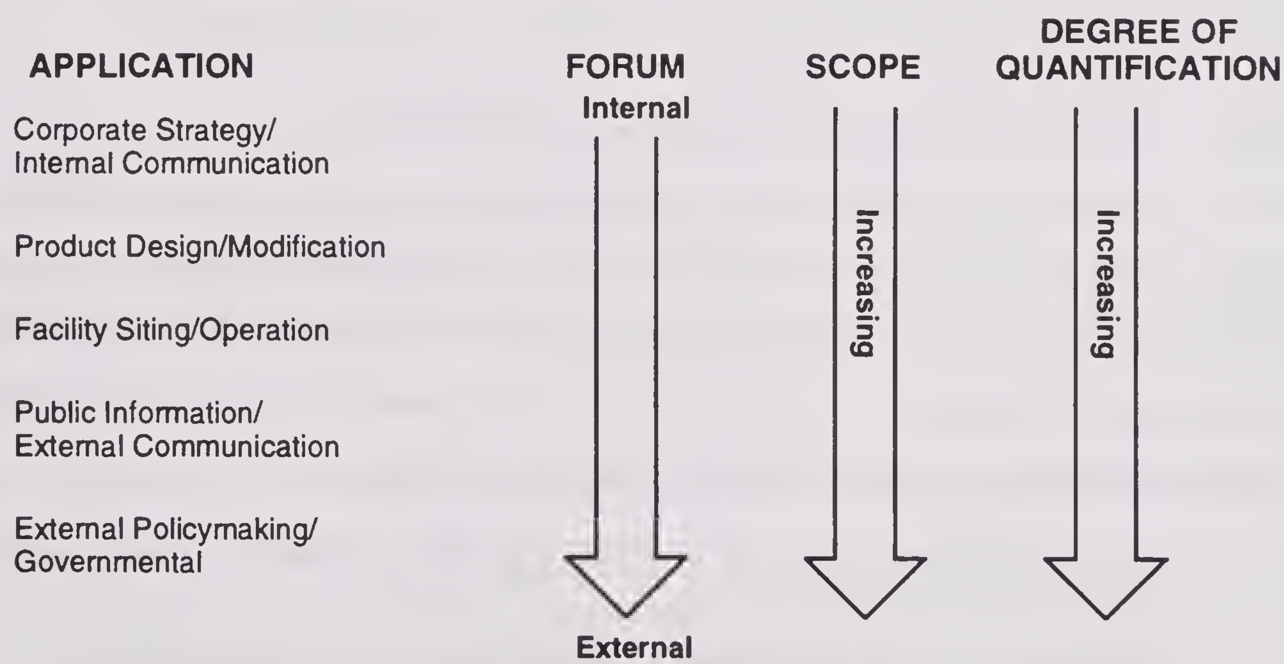


Figure 1-2. Range of LCA Applications

Source: SETAC, 1993

TABLE 1-2. POTENTIAL INTERNAL AND EXTERNAL APPLICATIONS FOR IMPACT ASSESSMENT

Internal Applications	External Applications
• Reduce future regulatory liability. • Compare impacts of generic or raw materials. • Identify materials, processes, or systems that create significant impacts. • Help develop long-term corporate policy regarding overall material use, resource conservation, and reduction of environmental impacts and risks. • Forecast potential impacts of new products or processes. • Compare alternatives within a particular process with the objective of minimizing impacts. • Aid in training designers in the use of lower impact product materials. • Internally evaluate impacts associated with source reduction and alternative waste management techniques. • Assess industrial process efficiency.	• Provide information that allows consumers or institutional buyers to evaluate and differentiate between products. • Provide information to policy makers, professional organizations, public-interest groups, and the general public about the environmental and human health consequences associated with a particular product or process life cycle, the use and release characteristics associated with a particular product or process life cycle, and potential impacts avoided by source reduction and alternative waste management techniques. • Help develop local, regional, or national long-term policy regarding overall material use, waste management, resource conservation, and reduction of environmental impacts and risks. • Supply information needed for legislative or regulatory policy that restricts or promotes the use of specific products, materials, or processes.

1.3.1 Internal Applications

An internal application of impact assessment is one in which results are never intended to be, and are never, released to the public (EPA, 1992a). An organization may conduct such an impact assessment, for example, to determine which production process exposes the organization to the least current and future regulatory liability.

For internal impact assessments, the sponsoring organization is not required to justify the methods, data sources, and items included and/or excluded outside the organization. Within the organization, these aspects of the LCA may or may not require as rigorous a justification as needed for an external application. While internal applications may not be required to follow stringent LCA guidelines, they should nonetheless follow the best practice. However, if the study results may be used externally at a later date, consideration should be given to conducting the assessment in the same manner as an external study.

Using a holistic, systematic approach to impact assessment that considers decisionmaking factors that were once outside the corporate sphere may significantly alter the corporate decisionmaking process. Corporations may find that performing an impact assessment is in their best interest because it may lead to impact reduction through waste minimization, more efficient production processes, and bottom-line cost savings.

1.3.2 External Applications

An external application of an impact assessment is one in which results are made available outside the sponsoring organization (EPA, 1992a). External impact assessment results may require more rigorous justification because they are open to additional scrutiny and thus may be faced with more intense peer review and disclosure of methods and results.

The public may expect external impact assessments to abide by impact assessment guidelines that represent a wide consensus of opinion. If guidelines are not followed, the public may request the sponsoring organization to provide information regarding the deviation from those guidelines. This request may be the case when the impact assessment results are used to support marketing claims that make product comparisons and may significantly affect other external entities, *or* when the results might affect public policy.

1.4 CURRENT STATE OF IMPACT ASSESSMENT PRACTICE

World Wildlife Fund (1991) recently updated a survey of three LCA practitioners—Battelle, Franklin Associates, Ltd., and Tellus Institute—to provide an overview of the state of impact assessment practice. Table 1-3 reports some of these results and describes the types of environmental and human health analyses currently performed by these three recognized LCA practitioners, as well as various methodological approaches used in impact assessment.

In addition, an industry survey by Sullivan and Ehrenfeld (1992) explored several companies' uses of analytic tools and programs designed to account for impacts throughout a product's life cycle. The survey revealed that the environmental impacts and life-cycle stages addressed by companies were fairly consistent. Air, water, soil emissions, and solid waste generation were addressed by all companies surveyed, and natural resource and energy use were addressed by eight of ten life-cycle frameworks. Habitat alteration was addressed by four of the ten frameworks, but biodiversity was rarely addressed.

The survey also found that, although many of the impact assessment frameworks used included elements that demonstrate life-cycle thinking, these frameworks are not standardized. Instead, they range from quantitative assessment techniques (e.g., indexing the importance of

various impacts) to more subjective techniques, such as consensus building and professional judgment (Sullivan and Ehrenfeld, 1992).

TABLE 1-3. PRACTITIONER SURVEY OF IMPACT ASSESSMENT CONSIDERATIONS

Considerations	Battelle	Franklin Associates, Ltd.	Tellus Institute
Amount/volume	Yes	Yes	Yes
Toxicity	Yes	Yes	Yes
Exposure	Only where generic pathway is defined.	Yes	No
Persistence	Via mechanical breakdown or degradability.	Yes	No
Mobility	Via surrogate measures (e.g., water solubility).	Yes	No
Global effects (e.g., climate change, ozone depletion)	Establish equivalency of various individual contributions.	Yes	Yes
Risk assessments	No	No	No
Consumer/worker safety	No	No	No
1. For releases to the environment, what criteria are used to select pollutants to measure			
a) pollutants covered by federal/state laws and regulations	Yes	Yes	Yes
b) pollutants that exceed some threshold level, regardless of regulatory controls	Yes	Yes	No
c) impact of pollutants (e.g., toxicity, etc.)	Establish impact potential networks (inventory vs. impacts).	Variable	Yes
d) SARA 313 list of toxic chemicals	—	—	—
2. Are releases assumed to meet current treatment standards?	Sometimes. Prefer actual releases; treatment standards used only if no other data are available.	Only if actual emission data are not available.	Only if actual emission data are not available.

(continued)

TABLE 1-3. PRACTITIONER SURVEY OF IMPACT ASSESSMENT CONSIDERATIONS (CONTINUED)

Considerations	Battelle	Franklin Associates, Ltd.	Tellus Institute
3. Is impact of individual pollutants estimated?	Try to assess whether concentration may be a problem only where a defined pathway and threshold level exist.	At least through the characterization phase.	Yes
4. What about relative impacts within and across media?	Have used valuation by Analytic Hierarchy Process (AHP) in streamlined LCA, but never in a conventional LCA.	Where comparison measures can be developed.	Methods developed to rank relative impacts, especially within media.
5. Is analysis primarily quantitative or qualitative?	Mix of qualitative/ quantitative depends on product stage and environmental pathways.	Mix—depending on the quality of data.	Quantitative

Source: Updated in 1994 from World Wildlife Fund, 1991.

1.5 SCOPE OF THIS DOCUMENT

This document outlines a conceptual framework, discusses key issues, and summarizes existing impact assessment methods. Chapter 2 discusses key issues related to the current use and future development of impact assessment. These issues include, but are not limited to, standardization of the impact assessment framework, scoping, uncertainty, data quality, value judgments, transparency, expert review, and presentation of results.

Chapter 3 outlines a conceptual framework for impact assessment, which includes three phases: classification of inventory items into impact categories, characterization of selected impacts, and valuation of impacts within and between impact categories. This chapter also discusses the different levels of analysis in the characterization phase, from less detailed loading assessment to more detailed risk assessment.

Chapters 4 through 7 summarize existing methods that have been presented, discussed, or used in the context of impact assessment. Chapter 4 profiles existing methods for characterizing impacts to ecosystems, human health, and natural resources. Chapter 5 discusses issues related to resource depletion and describes some existing methods for characterizing resource depletion. Chapter 6 presents those methods that apply to the valuation phase of impact assessment. Chapter 7 profiles integrated approaches that combine two or more phases of impact

assessment, typically the characterization and valuation phases. Chapter 8 reiterates key points regarding impact assessment and discusses potential future research needs to fill gaps in existing impact assessment procedures and methods as well as to better define the overall role of impact assessment in the LCA process.

Procedures and experience learned from environmental impact assessment as defined by the National Environmental Policy Act (NEPA) are included in Appendix A. Untested methods potentially useful for impact assessment are profiled in Appendix B. Appendix C provides key terms and definitions, and Appendix D is a bibliography of LCA-related literature.

CHAPTER 2

KEY ISSUES SURROUNDING IMPACT ASSESSMENT

Much of the current focus in the development of impact assessment is determining how to apply a wide variety of possible tools and methods within the impact assessment framework. This chapter discusses key issues related to the future development of impact assessment, including, but not limited to, standardization of the impact assessment framework, scoping, uncertainty, data quality, value judgments, transparency, expert review, and presentation of results.

2.1 STANDARDIZATION OF THE IMPACT ASSESSMENT FRAMEWORK

Increasing use of LCAs has resulted in a broad recognition that some degree of standardization of methodology is necessary to increase replicability and comparability, as well as public and peer confidence in external LCA studies (Denison, 1992b). To develop a standardized impact assessment framework, practitioners must agree not only on the conceptual aspects of impact assessment but also on other procedural aspects of impact assessment as well. These aspects might include

- a set of steps for the impact assessment practitioner to follow,
- a standardized list or checklist of impacts for the practitioner to consider,
- a common format for peer/expert review activities,
- a code of good practice for impact assessment as part of overall LCA studies, and
- a standardized presentation format for impact assessment results.

However, the question remains: Is it possible, and desirable, to develop a standardized impact assessment framework, or should the choice of framework be left to the practitioner? Although leaving the choice of impact assessment framework to the practitioner may be amenable for a wide variety of study scenarios and circumstances, using a standardized impact assessment framework could provide the following benefits:

- Users could make relative comparisons of studies without having to translate LCA studies to a common denominator.
- Potential misuse of impact assessment results to achieve a particular purpose or goal would be controlled.
- Practitioners would have objective guidelines for conducting an impact assessment.

- A standard listing of impact categories would remove some subjectivity in selecting impacts and would facilitate comparison between studies.
- Analysts would be able to incorporate results of other external LCAs into their studies.
- A consistent set of inventory and impact assessment data would be available to all interested parties.

On the other hand, a standardized framework for impact assessment may have little effect on LCA practices. For example, despite the development of guidelines for conducting inventory analysis, a number of significant discrepancies still exist in life-cycle inventory studies. Among other things, these discrepancies include differences in the definitions of the scope and process boundaries.

Only a few impact assessments have been conducted, and impact assessment procedures are still in their formative stages. Any standardized framework will undoubtedly be a function of future impact assessment research and experience. Therefore, researchers suggested using a phased approach in which an initial impact assessment framework is developed with presently available methods. Later, experience and insights derived from using the framework can be used to refine and/or develop new methods.

In this approach, experts develop and agree upon general principles and procedures, analysts begin preliminary case studies using these principles and procedures, and experts use feedback from preliminary studies to identify areas of need and to refine or redevelop the impact assessment process. Figure 2-1 provides a conceptual illustration of the phased-approach to impact assessment development.

A phased approach would allow researchers to use existing methods available for impact assessment while methods to fill gaps or analyze more difficult-to-determine impacts, such as habitat destruction, are developed. In addition, this approach would continue LCA and impact assessment case studies rather than delaying them in hopes of establishing the "ideal" approach.

2.2 USE OF SCOPING IN IMPACT ASSESSMENT

Scoping—deciding what will and will not be included in the study—is an integral part of LCA. In general, the impact assessment should consider all inputs and outputs compiled in the inventory. The assessment should also include justifiable reasons for any exclusions. Any justifiable reasons for any exclusions will be tied to the goals and scope of the LCA.

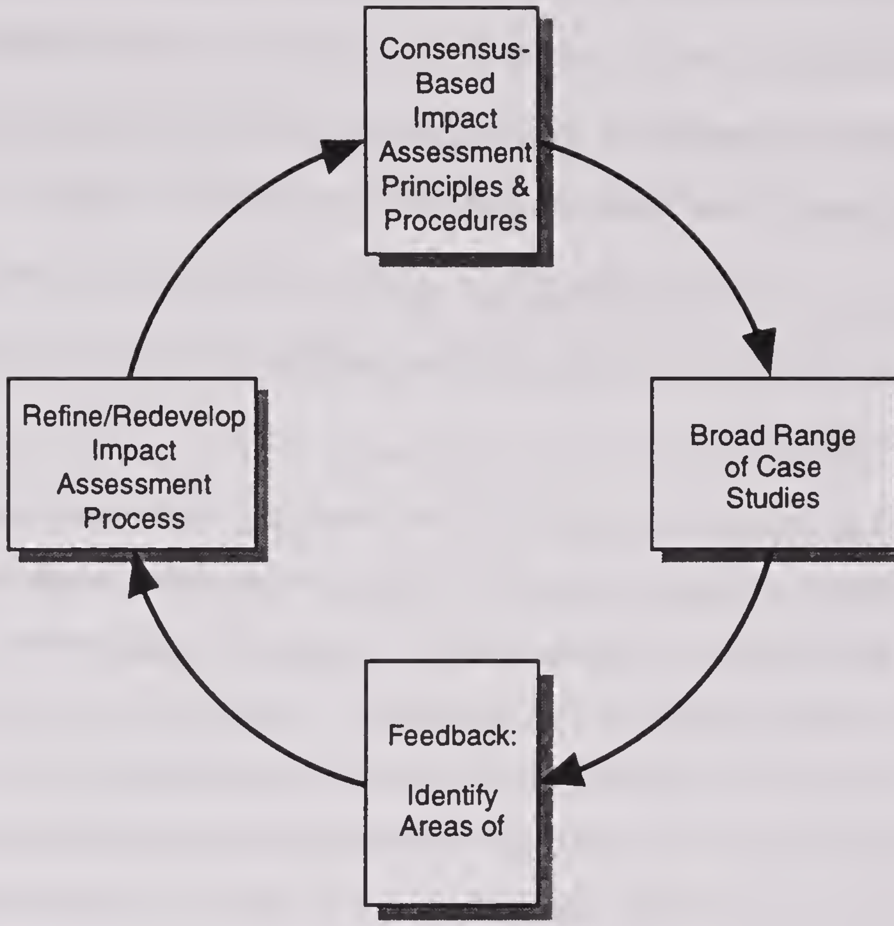


Figure 2-1. Phased Approach to Impact Assessment Development

On the other hand, in conducting an impact assessment, the practitioner may need to reevaluate the scope (to identify inventory items or impacts that will need additional data support), define which impacts are relevant to the LCA, and define the intended application or end use of the impact assessment results. Currently no set of rules exists that govern the type of information that can be used in an impact assessment, nor is there a clear need for one.

To define the scope of an impact assessment, the practitioner may find it useful to consider some primary scoping parameters specific to the impact assessment. These parameters might include the level of detail of the impact assessment, product system/potential impact boundaries, and the type of impact information required by decisionmakers. Some generic scoping parameters that are a function of the overall LCA include the following:

- matching the scope of the impact assessment to the goals and objectives of the LCA,
- identifying key inventory data that are missing or uncertain,
- identifying the variability of inventory data,
- identifying the impacts to be studied,
- recognizing the purpose(s) for conducting an impact assessment,
- determining how the results of the impact assessment are to be used,

- providing justification for excluding any elements,
- determining the level of impacts to be studied (source, media, or receptor),
- considering the audience to which results will be presented, and
- defining spatial and temporal boundaries (SETAC, 1993).

No one "correct scope" can be assigned to all impact assessments. The scope of an impact assessment will inevitably be a function of a number of study-specific variables such as goals, scope, and data limitations.

One important point to consider when initiating an impact assessment, or LCA in general, is the "nonthreshold assumption." The nonthreshold assumption simply says that no threshold exists for considering environmental loadings in an impact assessment. In other words, despite seemingly insignificant quantities, inventory items nonetheless contribute cumulatively to impacts and therefore may need to be considered in the impact assessment. For example, the energy used to manufacture a single automobile likely does not release enough SO_2 to the atmosphere to cause an appreciable rise in regional acid precipitation. However, when those SO_2 emissions are considered in the context of additional regional emissions, the SO_2 emissions may be considered a contributor to the regional acid precipitation and thus the SO_2 emissions may warrant consideration in the impact assessment.

The nonthreshold assumption may be of greater significance for some inventory items compared to others. For example, low concentrations of a noncarcinogenic, nonpersistent pollutant may be below a threshold of concern for human health effects. Practitioners may need to consider the appropriateness of the nonthreshold assumption for each inventory item with respect to the potential impact being assessed.

Incorporating the nonthreshold assumption into impact assessment not only provides justification for considering all inventory items in impact assessment, but also provides impetus for assessing the relative contribution of those inventory items to specific impact categories. In other words, the nonthreshold assumption makes it appropriately difficult for LCA practitioners to eliminate inventory items from further consideration on the basis that the quantity of inventory items is too insignificant to contribute to impacts.

One concern with using the nonthreshold assumption in the context of impact assessment is the possibility of misinterpreting the outcome of the impact assessment to represent actual causal association between inventory items and impacts. To avoid this situation, practitioners should make the use of the nonthreshold assumption transparent to users of the impact assessment. Section 2.8 discusses an approach for summarizing the results of an impact

assessment, providing a format to clearly communicate specific aspects of the assessment such as the nonthreshold assumption.

Scoping in impact assessment may draw in part from the scoping process required as part of Environmental Impact Assessment (EIA) by the U.S. Council on Environmental Quality (CEQ) in response to NEPA of 1969. The EIA scoping process is described in Appendix A.

2.3 COMMUNICATING UNCERTAINTY IN IMPACT ASSESSMENT

Determining fate and effects of pollutants and substances in the natural environment is extremely difficult. Uncertainty in the context of impact assessment extends beyond that in the inventory analysis. Inventory data usually are based on many assumptions, represent aggregated or averaged measures, contain many gaps, and are broad in nature (e.g., data from different plants with different levels of technology). Nonetheless these data typically are measurable inputs and outputs that can often times be bounded with some type of measure (e.g., range) of uncertainty. Because inventory data are the primary inputs for impact assessment, the range of uncertainty associated with the impact assessment model is partly dependent on the range of uncertainty associated with the inventory data. However, additional uncertainties are introduced in the impact assessment stage of an LCA. Section 2.3.3 discusses specific methods for uncertainty analysis. Practitioners should describe and discuss the uncertainties associated with LCA impact assessment methods, data, and results.

2.3.1 Translating Inventory Items to Impacts

A primary issue in impact assessment is the uncertainty surrounding the linking of inventory items to impacts. Scientific information may indicate that various inventory items are associated with, or have been shown to cause particular effects. However, it is difficult (if not impossible) to prove that a specific input or output from a specific LCA causes an actual effect. Thus the results of the impact assessment will likely not prove that the product system under consideration actually caused such impacts. None the less, a link can often be made between an inventory item and a potential impact, or multiple impacts. For example, SO_2 emissions have been linked to the formation of acid precipitation, which in turn can lead to other impacts such as tree damage, acidification of lakes, corrosion of buildings and materials, and the leaching of heavy metals from soils.

The causal uncertainty described above is primarily a result of limited understanding of such concepts as biochemical, physiological, and environmental interactions; fate and transport of substances released in an environmental setting; and the distribution of nonchemical stresses (e.g., heat, noise). Factors that may need to be considered to understand impact linkages include:

- the spatial and temporal scales of environmental loadings,
- interactions among human-induced loadings and natural loadings,
- natural variability and the problems of discriminating from "background noise," and
- the different modes of action of the loading on the environment (EPA, 1992d).

As a result of these factors, impact networks are often diverse, nonlinear, and largely site-specific, and involve a wide range of potential impacts at various thresholds. Although increased research in these areas may reduce a substantial amount of uncertainty, certain aspects of this uncertainty are intrinsically irreducible—for example, natural climatic variations among different locations. In addition, inventory data are generally *not* site-specific, which adds additional uncertainty to the analysis.

Clearly, a considerable array of complexities and uncertainties exist when translating inventory into potential impacts. A key issue is how to account for and communicate this uncertainty in the context of impact assessment or in the impact networks themselves. Some possible ways of incorporating uncertainty into impact networks include quantitative approaches such as incorporating probabilities or measures of compounded uncertainty into the linkages between inventory items and potential impacts and qualitative approaches such as using a set of qualitative evaluative criteria.

Quantitative Approaches

Some possible ways of incorporating uncertainty into inventory-to-impact links or impact-to-impact links include, among other things, incorporating probabilities or measures of compounded uncertainty into the links. In doing so, the practitioner must bear in mind that the two concepts have largely different effects on the expression of causal association, as described below:

1. **Joint Probability:** When given two events, A and B, the probability of both A and B occurring is the product of the probability of occurrence of A times the conditional probability of event B occurring (i.e., the probability of event B occurring given that event A has occurred). For example, consider the case where life-cycle inventory item X leads to potential impact A with a probability of 0.5, and potential impact A leads to potential impact B with a probability of 0.5 (none of these statements involves uncertainty). One can then state, on the basis of joint probability, that the probability of life-cycle inventory item X leading to potential impact B is 0.25 (0.5 x 0.5), and uncertainty plays no role.
2. **Compounded Uncertainty:** Using the above format, when given two events, A and B, each with a given range of uncertainty, the likelihood of both A and B occurring is the products of the ranges of uncertainty. For example, consider again

the case where life-cycle inventory item X leads to potential impact A with a range of probability of 0.2 to 0.8, and potential impact A leads to potential impact B with a range of probability of 0.2 to 0.8. The effect of compounded uncertainty is that one can only say that the likelihood of life-cycle inventory item X leading to potential impact B is between 0.04 and 0.64 (0.2 x 0.2 and 0.8 x 0.8).

The above examples illustrate a key distinction between joint probability and compounded uncertainty. Unlike joint probability, compounded uncertainty does not make further potential impacts less likely to occur but instead makes them increasingly more difficult to predict. This key difference should be kept in mind if either of these two methods are used as expressions of causal association.

In many impact assessments practitioners would not likely develop quantitative measures of joint probability or compounded uncertainty for relating inventory items to impacts but rather would express the inventory data as means and variances or ranges.

Qualitative Approaches

A possible qualitative approach to evaluating causal relationships among inventory items and impacts in an impact assessment is to use a set of evaluative criteria, such as those suggested by Hill (1965):

- strength (a high magnitude of impact is associated with a particular loading)
- consistency (the association is repeatedly observed under different circumstances)
- specificity (the impact is diagnostic of a loading)
- temporality (the loading precedes the impact in time)
- presence of a biological gradient (a positive correlation between loading and impact)
- a plausible mechanism of action
- coherence (the hypothesis does not conflict with knowledge of natural history)
- experimental evidence
- analogy (similar loadings cause similar impacts)

Although not all of these criteria need be satisfied to support causal association, each will incrementally reinforce the argument for causality. The presence of refuting evidence does not necessarily rule out causality. Instead, it may represent an incomplete understanding of the complex relationships at hand.

2.3.2 Impact Assessment Results

Even if impact assessment were able to provide a direct measure of uncertainty, the concern remains about compounding uncertainties from the inventory with uncertainties from an impact assessment. That is, when the LCA practitioner uses inventory data with much uncertainty associated with it and then incorporates the uncertainties associated with the impact assessment process described above, the resulting information is questionable in the decisionmaking process. Uncertainties surrounding different components of the environmental impacts evaluation affect the analyst's confidence in making a specific conclusion (EPA, 1994b).

Uncertainty clearly plays a large role in impact assessment. This is not to say that LCA or impact assessment has no use as a decisionmaking tool, because all decisionmaking tools contain some degree of uncertainty whether or not it is explicit. For example, similar problems exist in the field of risk assessment where analysts are constantly faced with data-input limitations and exposure and effect uncertainties. Like risk assessments, impact assessment can be performed with all levels of information, from abysmal to excellent, and can address a variety of levels of assessment, from release of individual substances to an environmental media to the release of multiple substances to multiple environmental media.

The key point is that the lower the quality of the information and models used in the assessment, the more uncertain the outcome. Therefore, developing a method of identifying and communicating uncertainty should benefit the users of impact assessment. Some general areas of impact assessment that may be used as indicators of the overall certainty or uncertainty of the assessment, and thus affect the quality and usefulness of results, might include the following:

- quality of input data,
- structure of impact characterization model,
- type of model testing, and
- level of expert review.

2.3.3 Methods of Uncertainty and Sensitivity Analysis

Both quantitative and qualitative techniques are available for expressing data quality in the context of impact assessment. These methods include the following:

Quantitative Methods

- confidence interval/data variability estimation
- accuracy, precision, and degree of bias measurement
- goodness of fit evaluation
- sensitivity analysis
- uncertainty analysis

Qualitative Methods

- limitations of life-cycle inventory data for predicting impacts
- validity, accuracy, and limitations of classifying inventory items into impact categories
- validity, accuracy, and limitations of conversion models used

Techniques such as sensitivity analysis or uncertainty analysis may provide useful starting points for impact assessment. Sensitivity analysis is a systematic procedure for estimating the effects of data uncertainties on the outcome of a computational model (EPA, 1993a). It provides a means of determining what does and does not matter in a computational model. Researchers have recognized that applying sensitivity analysis in the context of impact assessment may be useful in theory only, because it requires the quite difficult process of developing mathematical models to evaluate system parameters (EPA, 1994b). However, analysts may be able to develop variations of sensitivity analysis methods that better fit the needs of impact assessment. Table 2-1 describes proposed methods for sensitivity analysis in the context of impact assessment. For further discussions of these methods and their potential future applicability to LCA, the reader is referred to EPA (1992b).

Uncertainty analysis identifies, discusses, and quantifies, to the extent possible, the uncertainty in identifying and characterizing potential impacts. The total uncertainty in the LCA represents cumulative uncertainties from each phase of impact assessment. Using uncertainty analysis, a practitioner can evaluate the effect of uncertainties on the overall impact assessment and, when applicable, determine ways for reducing uncertainty. In addition to providing the practitioner with insight to the impact assessment's strengths and weaknesses, uncertainty analysis can also be used as a basis for decisionmaking purposes among comparative assessments.

Table 2-2 lists various methods for uncertainty analysis and their advantages. For further discussions of the use of these methods in the context of LCA, the reader is directed to EPA (1992b).

TABLE 2-1. POSSIBLE APPROACHES, ADVANTAGES, AND DISADVANTAGES OF SENSITIVITY ANALYSIS METHODS FOR IMPACT ASSESSMENT

Method	Advantages	Disadvantages
Tornado diagrams	• relatively simple to use • wide range of applicability	• requires the development of a mathematical model
Dominance considerations	• useful for determining the dominance of specific alternatives	• more applicable for option selection than for sensitivity evaluation
Two-way and three-way sensitivity analysis	• allows for evaluations of multiple variables at the same time • useful for evaluating impacts of alternatives	• does not focus on data quality per se
Deterministic sensitivity analysis	• applicable to LCA data-quality evaluation • identifies the most significant variables	• requires the development of a mathematical model

2.4 DATA AVAILABILITY AND QUALITY CONCERNS IN IMPACT ASSESSMENT

Recent LCA forums (SETAC LCA Data Quality Workshop, Wintergreen, Virginia, October 4-9, 1992; and SETAC Data Quality Open Forum, Washington, D.C., February 18, 1993) recognized that data quality is an integral component of the LCA process. LCA must be able to accommodate varying degrees of data availability, data types, and data quality. Because of the multiple and significant ways in which LCA information can be used, identifying and evaluating data quality and their relationship to LCA methodology are important. The following discussion of data quality issues focuses only on issues that are more specific to impact assessment.

EPA (1994b) has recently developed guidelines to aid LCA practitioners in assessing the quality of data used in inventory analyses. Data quality is defined in this document as the degree of confidence an analyst has in a data source or a data value based on defined data quality goals, data quality indicators, and the role of data quality in the overall context of the LCA (EPA, 1994b). These guidelines provide a framework for integrating data quality assessment into the inventory analysis process.

TABLE 2-2. POSSIBLE APPROACHES, ADVANTAGES, AND DISADVANTAGES OF UNCERTAINTY METHODS FOR IMPACT ASSESSMENT

Method	Advantages	Disadvantages
Analytic	• ranks contributors to uncertainty	• limited applicability
Monte Carlo	• economical • widely applicable • facilitates understanding of sampling distribution concepts	• sensitivity to input assumptions hard to assess • no ranking of uncertainty contributors • dependence on accurate information and covariance of input parameters
Response surfaces	• economical • widely applicable • ranking of uncertainty contributors	• accuracy hard to assess
Differential sensitivity	• widely applicable • ranks contributors to uncertainty	• long computation times possible • large model and code development costs
Evaluation of confidence intervals	• measures uncertainty due to statistical variability in data	• limited applicability • no ranking of uncertainty contributors

Source: Cox and Baybutt, 1981.

Similar to the framework discussed above, a data quality assessment framework is needed to integrate data quality assessment into the impact assessment process. Currently no protocol has been developed for assessing the quality of data in impact assessments. In addition to the quality of data received from inventory, practitioners must also consider the quality of additional data (e.g., toxicity, bioaccumulation, assimilation, equivalency factors, etc.) needed to conduct an impact assessment.

The purpose of this section is to outline some of the significant data quality issues facing impact assessment. With very few impact studies to draw from, pinpointing all the data quality issues that will be integral to impact assessment is very difficult. However, a recent Tellus Institute analysis of impacts associated with the production and disposal of packaging materials found basic problems that were related to data used as input parameters for impact analysis, including the following:

- A lack of systematic data on some components of the product system limited the scope of the analysis as well as the modeling of significant processes or activities.
- Publicly available databases often contained out-of-date data (Tellus Institute, 1992a).

2.4.1 Evaluating Data Availability

Although the lack of available data required for impact assessment is a primary concern, many sources of data may be useful for conducting an LCA. EPA (1994a) provides a comprehensive overview of publicly available data sources for conducting an LCA. Table 2-3 summarizes data needs for impact assessment in terms of a five-tiered system of increasing data quality and decreasing data availability.

TABLE 2-3. IMPACT ASSESSMENT DATA NEEDS

Conversion Model Tier	Data Needs
Tier 1: Loading Assessment	Mass, volume, or other units of physical quantity.
Tier 2: Equivalency Assessment	Same as Tier 1 plus equivalency algorithms based on hazard data. Also may include measures for resource stocks and yields, as well as nonchemical loadings.
Tier 3: Toxicity, Persistence, and Bioaccumulation	Same as Tier 1 and 2 plus information on interaction of chemicals with the environment (i.e., persistence and bioaccumulation) and toxicity data. Also may include measures for resource stocks and yields, as well as nonchemical loadings.
Tier 4: Generic Exposure/Effects Assessment	Same as Tier 1 plus generic environmental and human health data.
Tier 5: Site-Specific Exposure/ Effects Assessment	Same as Tier 1 plus site-specific exposure and environmental and human health data.

Source: SETAC, 1993

A recent SETAC-sponsored LCA Data Quality Workshop in Wintergreen, Virginia, recognized that currently available environmental input and output data can only support Tier 2- to Tier 3-type models. Although many feel that such a method can be improved, others have recognized the lack of information for Tier 2 to Tier 5 conversion models.

Advancing to Tier 2- and Tier 3-type conversion models, which require equivalency factors and chemical properties data, will require the inventory to contain an increased level of

chemical and site specificity. However, such a level of data quality may be achievable in the near term. Tier 4-type conversion models may use the same data as Tier 3-type models, only in a different manner. However, in general, to move to Tier 4- and Tier 5-type conversion models, process- or activity-specific, unaggregated and unaveraged inventory data will be needed. Inventory data are currently unable to support such models.

In the near term, researchers may be able to develop a database of information that is specifically designed for use in LCA. It would contain a variety of information on basic commodities and pollutants that serve as inputs and outputs to many product or process life cycles, respectively. Such a database could serve as a clearinghouse for generic information for supporting LCAs and other types of residuals-based analyses.

2.4.2 Evaluating Data Quality

As stated in the beginning of this section, data quality in the context of LCA is defined as the degree of confidence an analyst has in a data source or a data value (EPA, 1994b). A primary concern with respect to data quality in this context is the use of less-than-perfect inventory data in less-than-perfect impact assessment models as described in the previous section on uncertainty. The resulting information may have questionable usefulness for decisionmaking purposes. For example, aggregated secondary data are typically used in inventories. Aggregated data are not useful for some impact assessment methods, such as fate and transport models or exposure assessment, that require site-specific data (EPA, 1994b).

A second concern in impact assessment is the quality of additional data needed by conversion models. The only type of conversion model that does not require any additional data is loading assessment, where inventory data are used directly (see Table 2-1). Any model beyond loading assessment requires additional information such as toxicity, persistence, bioaccumulation, and equivalency factors. Even with the highest quality inventory data, impact assessment results can be compromised if low quality information (e.g., environmental characteristics, toxicity measures) is used in the conversion models.

A third concern is the quality of the conversion models themselves. That is, even with perfect input information, the quality of impact assessment results is governed by the predictive accuracy of the conversion model(s) used. This issue is not limited to impact assessment but includes any type of analysis that employs models to transform data into more useful and meaningful forms. At this stage of impact assessment development, conversion models need to

be developed and validated as well. Therefore, one of the goals of impact assessment may be to make the limitations of conversion models, as well as any additional information required by the models, transparent to the users of the results.

2.5 INCORPORATING VALUE JUDGMENTS INTO IMPACT ASSESSMENT

Impact assessment is similar to other decisionmaking support systems in that it involves applying subjective value judgments. Nothing is inherently right or wrong with value judgments. All individuals and institutions have subjective values, which they express either explicitly or implicitly. A key objective of impact assessment is to make subjective value judgments transparent so that users and others will know the basis from which the assessment was conducted and any conclusions drawn. Clearly articulating subjective value judgments lets the user know the values that guided the impact assessment.

Both the practitioner and the user of the resulting impact assessment make value judgments as the result of such considerations as the presence of uncertainties, data limitations, and impact assessment model limitations. Because different individuals and institutions have different values, there is no "correct" set of values to use during the impact assessment. However, for purposes of impact assessment, and LCA in general, developing a standard protocol for identifying and evaluating value judgments may be worthwhile.

In the face of incomplete information and uncertain cause-and-effect relationships, the practitioner may need to make judgments based on the available evidence. The main problem in making value judgments about cause-and-effect relationships is that directly applicable data are often insufficient. In such a case, the practitioner must use value judgments to make the best possible assessment of the relationship given the information at hand. Furthermore, because the extrapolation of value judgments depends on the practitioner's interpretation of the impact assessment literature, different people will have different interpretations and, thus, different value judgments. Unlike other forms of uncertainty (e.g., measurement and sampling error) that can be generally calculated by means of standard procedures, the type of uncertainty described above cannot be directly quantified because of its judgmental nature.

Value judgments occur at varying degrees throughout the impact assessment process. Within the impact assessment component, value judgments can occur at any of the following points:

- goal definition and scoping
- classification of inventory items to impact categories
- determination of impacts of concern
- evaluation and selection of models to characterize impacts of concern
- interpretation of results obtained from impact characterization efforts
- development of assumptions based on logic and scientific principles to fill data gaps
- evaluation and selection of ranking or weighting schemes in the valuation phase

In summary, value judgments are integral parts of any decisionmaking system, including environmental policy decisions. In that regard, impact assessment is no different from any other public, private, or individual decision that can affect the environment or human health. It is recommended that practitioners clearly articulate those value judgments either qualitatively or quantitatively and discuss the scientific basis or evidence and any philosophical, cultural, or intellectual influences for making the judgments. Employing a method such as encoding probability judgments may provide a means of identifying and quantitatively characterizing value judgments. Also, this method would enable users of the impact assessment to understand the frame of reference from which the impact assessment was conducted, even though they may not personally agree with it.

2.6 TRANSPARENCY

Because assumptions and value judgments are integral parts of impact assessment and many other decisionmaking systems and they shouldn't be eliminated in LCAs, their use must be made transparent (i.e., clearly defined). Transparency entails full disclosure of the content and conduct of the impact assessment process, including assumptions and subjective value judgments. The practitioner should strive to present the following specific aspects of an impact assessment in a transparent manner:

- goals of the LCA and impact assessment
- scope and boundary settings
- data sources/data quality/data variability—uncertainty
- models/methods used in the impact assessment process
  - assumptions
  - limitations
- data or methodology manipulations

- value judgments
- exclusions
- lost information due to aggregation, etc.
- analyst's interpretation of the implications of all above items on LCA results

Transparency in reporting impact assessment results is important for replicability. By fully disclosing all aspects of an impact assessment, as listed above, the practitioner enables an external observer or investigator to start with the same original data and reproduce the impact assessment results.

Although barriers to full disclosure (e.g., proprietary data, practitioners' self interest in keeping their methods or databases to themselves) clearly exist in LCA studies, practitioners should strive to make their studies as replicable as possible and should fully explain and justify factors that preclude them from doing so (Denison, 1992b).

Reproducibility is important to support the understanding and credibility of the impact assessment results. For example, a current working study comparing two existing LCAs of corrugated cardboard found that differing results were largely due to differences in study scope and boundary settings (Ekvall, 1992b).

2.7 EXPERT PEER REVIEW

Scientific data and methodologies used in impact assessment are based on information that is frequently complex, conflicting, ambiguous, or incomplete. Therefore, EPA supports the creation of an expert peer review process for impact assessment, and LCA in general, to advance the quality and consistency of LCAs. The desirability of an expert peer review process stems from four main areas of concern: 1) the lack of understanding of the scope or methodology used in LCAs, 2) the desire to verify data used and practitioner's compilation of data, 3) questioning of assumptions used and the overall results, and 4) the communication of results (EPA, 1993a).

Practitioners can evaluate the viability and accuracy of impact assessments by establishing an expert peer review process for impact assessment. Expert peer review can be integrated into the following stages of impact assessment:

- determining the purpose and scope of the impact assessment;
- evaluating data sources and the quality of data used in the assessment;
- evaluating and selecting assessment and measurement endpoints;
- evaluating and selecting conversion models;

- developing assumptions, etc., to fill data gaps; and
- interpreting and presenting impact assessment results.

The heightened recognition of the importance and necessity of expert peer review in LCA studies prompted SETAC to develop an interim expert peer review framework. This interim framework consists of four main steps:

1. Identify and assemble an expert peer review panel based on specified criteria.
2. Review the purpose, study boundaries, and databases of the LCA.
3. Review the stand-alone data compiled in the life-cycle inventory.
4. Review the draft final report (SETAC, 1992).

The purpose of this discussion on expert peer review is not to recommend a specific approach for an expert peer review process but rather to identify the reasons for having an expert peer review process for impact assessment and to discuss some issues for consideration before establishing an expert peer review protocol.

A related issue is how to conduct these expert peer reviews. For example,

- Should the expert peer review process use a standard checklist of review items?
- What is the appropriate timing of the expert peer review process with respect to conducting the impact assessment?
- Who should pay for the review with respect to internal and external applications?
- How should the expert peer review panel members be chosen?

Because impact assessment is in its developmental stages and involves many concepts and methods that have yet to be corroborated in practice, the use of an expert peer review panel will be a key role in shaping the future of impact assessment. The expert peer review panel is foreseen to consist of a relatively small but diverse group of individuals with experience using impact assessment methods and/or technical LCA procedures. In addition, although expert peer review is a critical component of both internal and external applications of impact assessment, a more stringent level of expert peer review will be required for external applications.

2.8 PRESENTATION OF IMPACT ASSESSMENT RESULTS

One of the more important aspects of impact assessment is the manner in which results are presented to the intended audience. The results of an impact assessment need to be presented in an effective manner that facilitates the decisionmaking process. Too much information of too many different types can result in information overload, whereas too little can hinder the

decisionmaking process. Practitioners should strive to conduct credible assessments and present the results objectively.

Specific aspects of the impact assessment that need to be documented and presented to decisionmakers include the following:

Content Aspects

- Clearly delineate scoping activities, including how the boundaries of analysis were determined.
- Report any objective data or results separately from subjective data or results.
- Express the characteristics of the database, including data sources, uncertainty, and assumptions.
- Clearly delineate analysis of actual versus potential impacts.

Conduct Aspects

- Provide justification for all impacts that were excluded from the analysis.
- Describe the use of assumptions, including how and by whom they were made.
- Describe the use of subjective value judgments, including how and by whom they were made.
- Describe any limitations and/or uncertainties of the valuation method.

Table 2-4 shows some possible methods for presenting impact assessment results and their corresponding advantages and disadvantages. A summary chart can be developed based on one of the methods described in Table 2-4 to present an overview of impact assessment results. The chart would provide a variety of information beyond the results of the impact assessment valuation process, such as

- a summary of the data used in the impact assessment including measures of variability/uncertainty,
- a summary of assumptions and value judgments made in the impact assessment,
- a description of the methods/models used in the impact assessment,
- a description of problems encountered and how they were resolved, and
- a summary of unresolved issues.

TABLE 2-4. COMPARISON OF IMPACT ASSESSMENT RESULTS PRESENTATION METHODS

Presentation Method	Objective	Advantages	Disadvantages
Single Score	• Provides a single impact score by aggregating all impacts based on a common denominator	• Provides comparative value • Easy to communicate	• Ambiguous derivation • Does not allow for a relative comparison of impacts • Difficult to incorporate qualitative data • Provides no information beyond valuation results • Fixes subjective values that cannot be shared by others.
Qualitative Rank	• Uses symbols or ranges of subjective impact scores to provide an overall ranking of impacts	• Provides comparative value • Easy to communicate	• Provides no quantitative support • Difficult to express relative comparison of impacts • Too simplistic • Provides no information beyond valuation results
Prioritization	• Prioritizes impacts based on subjective values that attempt to identify the more and less pressing impacts	• Provides comparative value • Identifies high- and low-priority items	• Encourages focus on only the top priorities • Priorities are often subjective in nature • Provides no information beyond valuation results
Impact Score Matrix	• Provides a quantitative or qualitative overview of impact categories	• Provides comparative value • Provides standard format • Incorporates quantitative and qualitative data	• Potential for confusion with too much information • Provides no information beyond valuation results
Impact Assessment Summary Chart	• Provides a variety of information beyond impact assessment results (such as databases used, methods used, value judgments, and limitations), which gives decisionmakers an overview of the impact assessment process	• Provides comparative value of entire impact assessment process • Provides information on impact assessment process beyond valuation results • Provides a standard format • Incorporates quantitative and qualitative data	• Potential for confusion with too much information

Decisionmakers could use this information as a tool to look at the overall picture or to focus on a particular aspect of the impact assessment. A standard presentation format would provide a clear and effective means of communicating the potentially complex array of information inherent in impact assessment or any impact analysis. Figure 2-2 provides a possible framework for the impact assessment summary chart. This kind of chart offers a relatively objective method of presentation where impact assessment results are presented in the context of the underlying data, methods, assumptions, and limitations used to achieve those results.

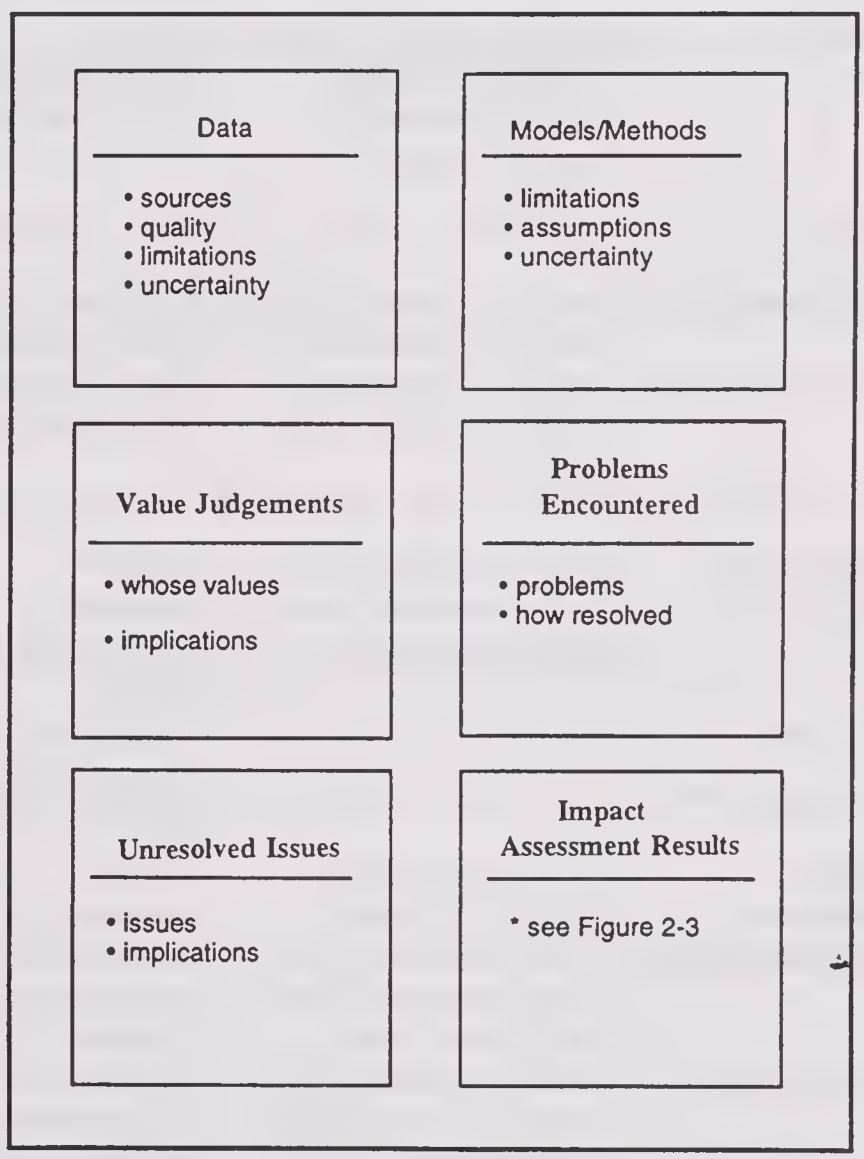


Figure 2-2. Impact Assessment Results Summary Chart

The impact assessment results portion of the summary chart could become complex and overwhelming because of the large amount of information and variety of different results. To help communicate the impact assessment results in a concise and comprehensible manner, a format can be developed for presenting impact assessment results within the overall impact assessment summary chart. Figure 2-3 illustrates a possible format for organizing impact assessment results based on the life-cycle stage and category in which the impact occurs.

	Life-Cycle Stage				
Impact	**Raw Materials Acquisition**	**Manufacturing Processing**	**Distribution/ Transportation**	**Use/Reuse/ Maintenance**	**Waste Management**
Ecosystem					
air					
water					
land					
biodiversity					
waste					
other					
Human Health					
occupational					
nonoccupational					
other					
Resource					
stock					
flow					
other					

Note: Several ranking methods could be used in this matrix:
> Pluses (+) for significant impact areas, Minuses (–) for less significant impact areas.
> ● = High, ◗ = Medium, ○ = Low.
> Numerical Ranking (e.g., 1 – 10) of the significance of impact areas.

Figure 2-3. Example of a Possible Impact Assessment Results Format

The impact assessment results table could handle both quantitative and qualitative information. Qualitative information could be expressed by symbols, such as + or -, indicating more or less significant impacts, or by circles with varied degrees of shading, indicating the magnitude of the impact.

2.9 UNRESOLVED ISSUES

Because impact assessment is still evolving, many issues persist that may play a role in the future development of impact assessment procedures and methods:

1. Although it is recognized that impact assessment is an inherently value-laden exercise, the following questions remain:
   - Who makes the value judgments?
   - Is it feasible to use external expert review for value judgments?
   - Should guidelines be required for value-laden areas, such as valuation, to help minimize the level of subjectivity in impact assessments?
2. To control potential misuse of impact assessments, quality standards may be needed when impact assessments are used for external purposes.
3. Specific evaluation methods (conversion models and impact descriptors) and valuation methods need to be chosen for analyzing specific impact categories.
4. Although methods such as risk assessment and fate and transport models can be used for impact assessment, analyzing multiple sites may be overly costly and impractical because of the requirement for additional data.
5. Much uncertainty persists in linking inventory items to impacts. Techniques are needed to estimate and integrate this uncertainty into impact assessment.
6. The political environment under which the LCA is conducted may affect the scope and impact considerations.
7. Practicality of a "cookbook" of impact assessment methods versus a more streamlined approach
8. Incorporation of economic impact (i.e., cost) information into impact assessment
9. Treatment of chronic versus acute impacts
10. Effects of impacts on future generations
11. Impact distribution equity considerations (e.g., impacts on children or other special subpopulations)
12. Treatment of human-induced versus naturally caused impacts

CHAPTER 3

A CONCEPTUAL FRAMEWORK FOR IMPACT ASSESSMENT

A major achievement of the SETAC-sponsored Life-Cycle Impact Analysis Workshop that took place in Sandestin, Florida, during February 1992, was the development of a three-phase conceptual framework for life-cycle impact assessment. This three-phase conceptual framework, illustrated in Figure 3-1, contains the following activities:

1. **Classification:** the process of assignment and initial aggregation of life-cycle inventory data to relatively homogeneous groupings of impacts (e.g., photochemical smog, lung disease, fossil-fuel depletion) within primary impact categories (e.g., ecosystem, human health, and natural resources).
2. **Characterization:** the qualitative and/or quantitative evaluation of potential impacts. The process of identifying impacts of concern (called assessment endpoints) and selecting actual or surrogate characteristics (called measurement endpoints) to describe the impacts. Characterization involves using specific impact assessment models to develop impact descriptors.
3. **Valuation:** the explicit and collective process of assigning relative values and/or weights to impacts using informal or formal valuation methods.

In Figure 3-1 the flow from the inventory analysis to improvement assessment is not necessarily linear because the sequence involves interrelationships and feedback loops among the major components. This is consistent with the three-component LCA triangle illustrated in Figure 1-1. For example, not only can opportunities for environmental and human health improvement be realized at any phase of the LCA, but unplanned modifications may entail revisiting previously completed components. Each LCA phase is discussed in detail in later chapters of this report.

Selecting the best-suited approach for conducting a particular impact assessment from a variety of available methods is important. Practitioners can use the following key decision points to help select the best-suited approach and shape the assessment:

- selecting the goals and scope of the study,
- learning stakeholder values and information needs, and
- characterizing the desired results.

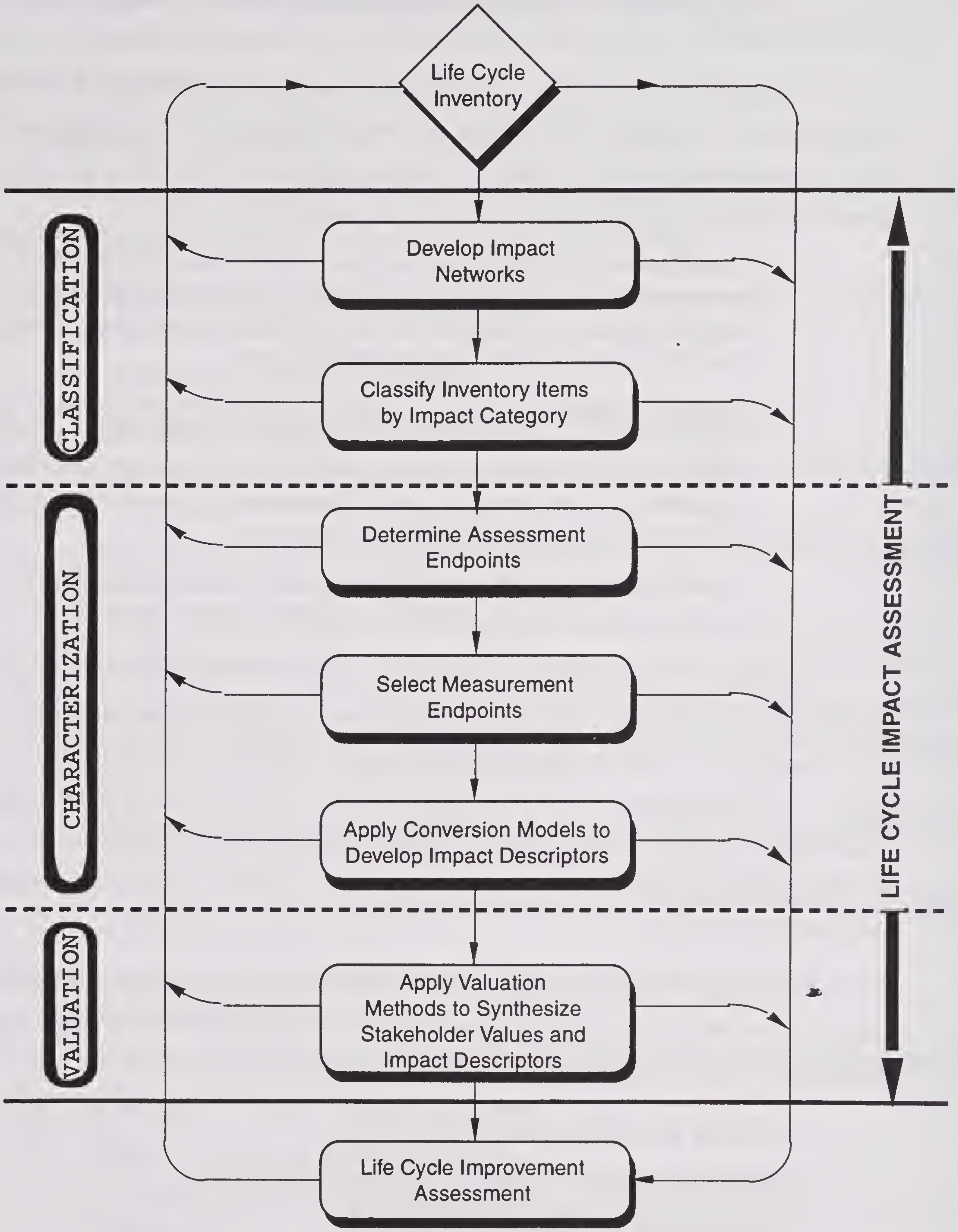


Figure 3-1. Conceptual Framework for Life-Cycle Impact Assessment

Figure 3-2 illustrates how these key LCA decision points fit in the LCA conceptual framework, emphasizing the impact assessment component. The decision points would be constantly revisited throughout the impact assessment and especially in the following impact assessment activities:

- classifying inventory items into impact categories;
- determining impacts, or categories of impacts, of concerns;
- choosing a model, or models, to characterize impacts; and
- valuing impacts, or categories of impacts.

3.1 CLASSIFICATION

When inventory items are taken from, or released to, the environment, they are considered potential causes of environmental and human health impacts. The classification phase of impact assessment provides a preliminary link between inventory items and potential impacts. The overall purpose of the classification phase is to organize and possibly aggregate inventory items into impact categories, which provide a more useful and manageable set of data. This process is accomplished through the two discrete activities of the classification phase:

- using existing or developing new impact networks to identify possible impacts associated with specific inventory items, and
- classifying inventory items within appropriate impact categories.

3.1.1 Developing Impact Networks

The preliminary activity of the classification phase of impact assessment is to *qualitatively* associate, or link, inventory items with subsequent impacts. This qualitative link can be established by reviewing each inventory item in the literature to determine its associated environmental impact(s). Further review of the literature can identify additional impacts that are associated with the initial impact. For example, consider that a quantity of SO_2 emissions released into the atmosphere is an item specified in the inventory analysis. A review of the impact assessment literature might identify the theory that SO_2 released into the atmosphere can lead to the formation of acid precipitation. Acid precipitation, in turn, can be found to lead to a number of additional impacts, such as the destruction of high-altitude forests, acidification of water bodies, corrosion of buildings and materials, and leaching of metals from soils. Further search of the impact assessment literature may reveal that these impacts can induce other identifiable impacts and so on.

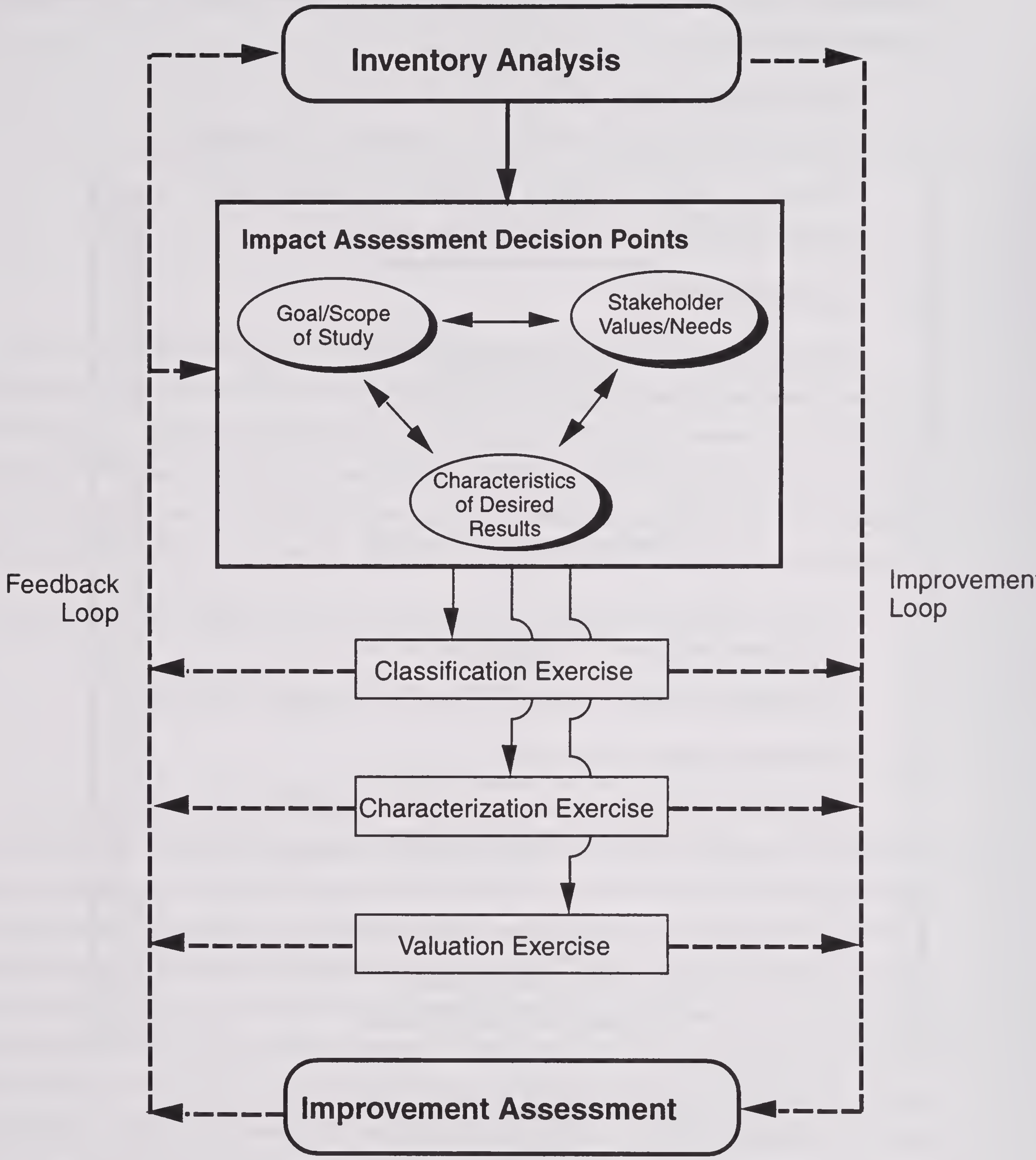


Figure 3-2. Key Impact Assessment Decision Points

Associating, or linking, inventory items to their respective impacts is a key issue of impact assessment because the pathways linking inventory items to their impacts typically are complex and nonlinear. Practitioners can use existing or develop new impact networks to aid in mapping out impact pathways. Networks of potential impacts are conceptual diagrams that illustrate qualitative links between inventory items and potential impacts. As the use of the term "qualitative links" implies, these networks do not necessarily provide a description of actual impacts. Instead, networks provide a means of identifying all the various *potential* impacts that can be associated with inventory items.

Consider the case of a given quantity of carbon dioxide (CO_2) identified in the inventory analysis. A search of the literature reveals that CO_2 is often linked to the greenhouse effect, which is a buildup of CO_2 and other gases that are relatively transparent to sunlight but trap heat by more efficiently absorbing the longer wave infrared radiation released by the earth (Schneider, 1990). In turn, an enhanced greenhouse effect is linked to other impacts such as global warming, which in turn is linked to regional climate change. This example, as well as the basic framework for building an impact network, is illustrated in Figure 3-3.

Developing impact networks can be a difficult task. Pathways from inventory items to impacts may not yet be fully identified and many factors govern how and what kind of impacts will result. Because many pathways and impacts can exist, tracing impact networks through a number of different pathways may be necessary.[1]

As an example of a multiple pathway impact network, consider the case of nitrogen oxides (NO_x) released from a coal-fired electric plant, as shown in Figure 3-4. In other situations, multiple inventory items can lead to a similar impact or impacts. As an example of such a scenario, consider the greenhouse effect.

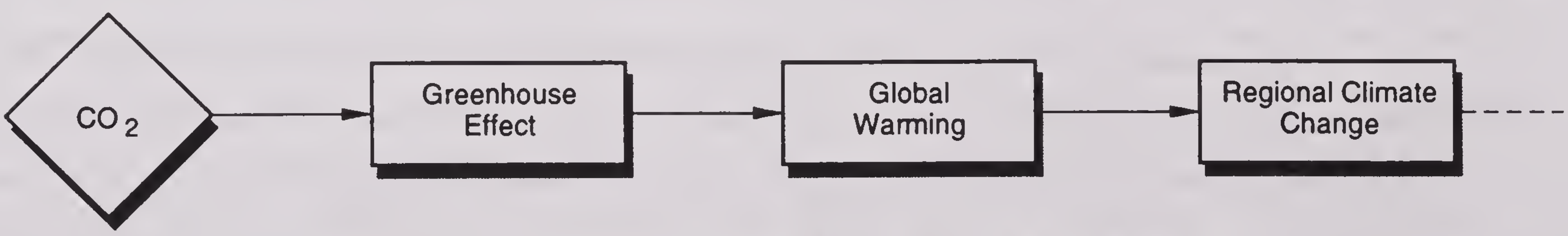


Figure 3-3. Example of Basic Network Using CO_2

[1] In this report we do not use the terms primary, secondary, or tertiary to distinguish impact levels because of the implicit valuation imbedded in those terms and the difficulty of assigning the terms to a complex web of impacts typical of many impact networks.

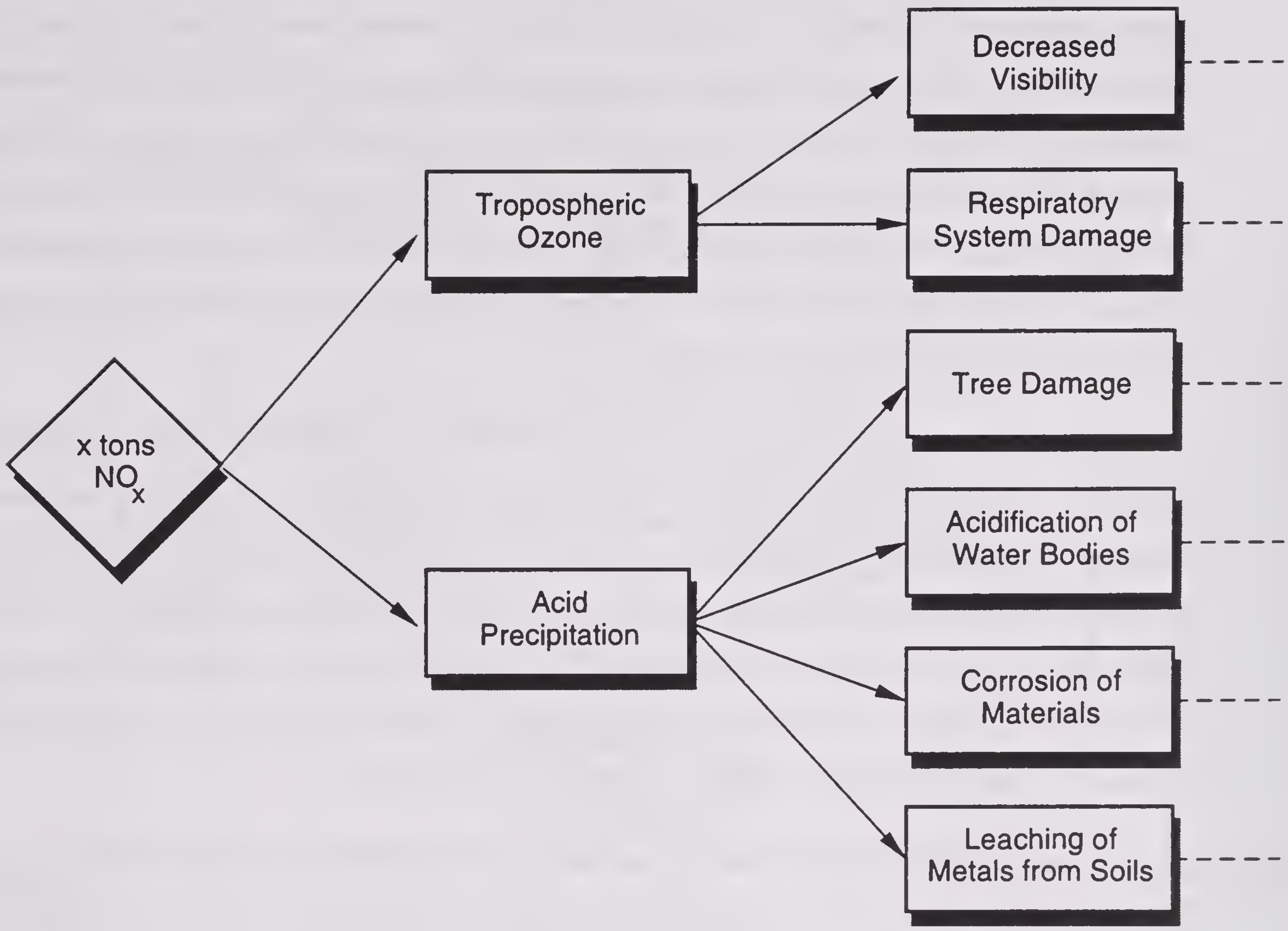


Figure 3-4. NO_x Example of Multiple Pathway Impact Network from a Single Inventory Item

Like many other impacts, a number of different substances may contribute to the greenhouse effect, as demonstrated in Figure 3-5.

In summary, linking inventory items to impacts can take a variety of forms. The linkage can range from a simple linear one (as shown in Figure 3-3) to one that involves linear and nonlinear relationships between multiple inventory items and multiple impacts (as shown in Figures 3-4 and 3-5). It is expected that a "library" of networks will be developed through the practice of impact assessment, making such assessments increasingly more feasible and economical.

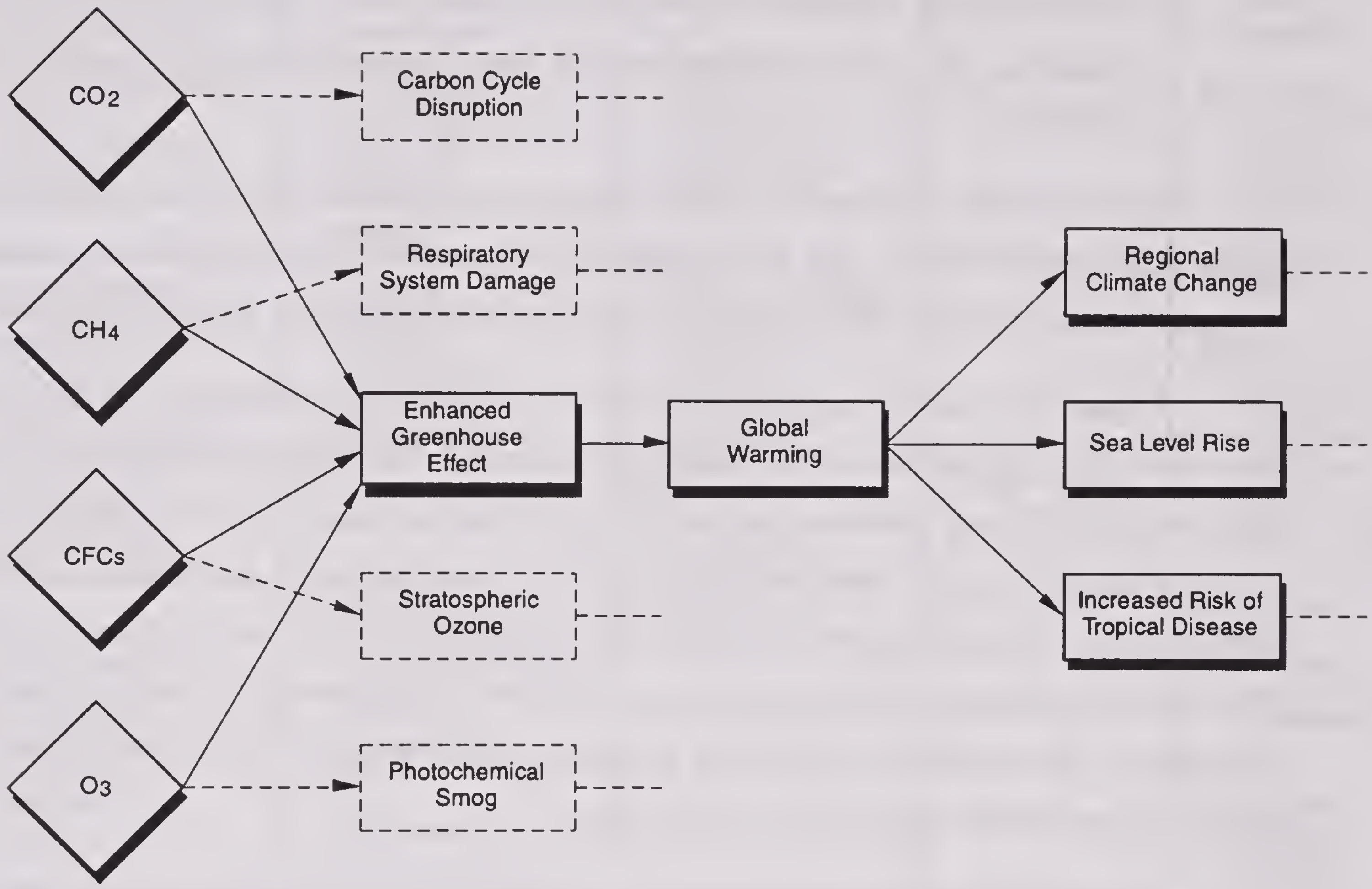


Figure 3-5. Example of Multiple Inventory Items Leading to Similar Impacts

3.1.2 Classifying Inventory Items Within Impact Categories

After referring to existing impact networks or developing new ones, practitioners should review the networks to see if they contain any inherent structure that enables them to establish a set of impact categories under which inventory items can be grouped. For example, during the review of impact networks, researchers might identify that quantities of air emissions listed in the inventory analysis, such as CO_2, methane (CH_4), chlorofluorocarbons (CFCs), and ozone (O_3), all contribute to the greenhouse effect. All four of these inventory items thus can be grouped in the subcategory, greenhouse effect, within the main ecosystem impacts category.

The three main categories of impacts considered in an impact assessment include impacts to ecosystems, human health, and natural resources. Social welfare may be considered as an additional impact category, although currently no tools yield a credible analysis of such impacts.

Despite the current lack of tools to analyze social welfare impacts, practitioners can attempt to incorporate social welfare impacts by

- identifying the impacts of a product or process life cycle on social welfare, and
- identifying the effect of social welfare impacts on ecosystems, human health, or natural resources.

As an example of a social welfare impact, consider the large labor force required to manufacture automobiles. The immigration of a large labor force into the area may result in impacts such as overcrowding and degradation of pristine habitat in nearby recreation areas.

Figure 3-6 provides a generic example of possible impact categories and subcategories as developed from a hypothetical set of impact networks. The suggested approach to classification is to first build impact networks and see if they contain any inherent structure for developing subcategories of impacts rather than starting with a prestructured, and value-laden, list of impact subcategories. This approach to classification is essentially the same as that used by SETAC (1993), which groups inventory items into relatively homogeneous problem types, called stressor categories. Our approach differs only in that the term "stressor" is not used because of ongoing confusion associated with the use of that term.

3.1.3 Example Classification Exercise of High-Density Polyethylene (HDPE) Production

An inventory analysis of an HDPE production system would likely include numerous components. Table 3-1 provides examples of information developed in an inventory analysis for the manufacture of HDPE.

Ecosystem, human health, and natural resource impacts associated with the items listed in Table 3-1 can be determined by searching the impact assessment literature. For example, the release of SO_2 from the manufacture of HDPE, as shown in Table 3-1, can be evaluated for potential impacts by searching the literature for the effects of SO_2 released into the atmosphere. From this search, it will likely be determined that SO_2 emissions to the atmosphere often combine with other atmospheric compounds to produce acid precipitation. Thus, as shown in Table 3-2, SO_2 can be categorized under the ecosystem impact category of acidification.

Ecosystem

Atmosphere
- toxicity impacts
- ozone depletion
- greenhouse effect
- visibility changes
- smog/fog
- ground-level ozone buildup
- climatic change
  - micro
  - macro

Water
- toxicity impacts
- contamination
- depletion
  - surface
  - ground
- thermal changes
- turbidity changes
- acidification
- nutrification
- eutrophication
- chemical alteration

Soil
- toxicity impacts
- salinity
- lateritization
- podzolization
- acidification
- fertilization
- erosion

Others
- geomorphic effects
- biodiversity effects
- habitat alterations
- animal welfare

Human Health

Chronic Effects
- carcinogenic
- mutagenic
- teratogenic
- neurological damage
- reproductive disorders
- major organ diseases
  - heart
  - lung
  - liver
  - kidney
- radiation
  - ionized
  - ultraviolet
  - heat
- physiological
- anemia
- skin disease
- sterility

Acute Effects
- accidents
  - occupational
  - nonoccupational
- radiation
  - ionized
  - ultraviolet
  - heat
- noise
- odor, taste, etc.
- microorganisms

Natural Resources

Stock
- fossil fuels
- minerals
- atomic energy
- soil
- space (e.g., landfill)
- atmosphere
- hydrosphere
- aesthetics of planet

Flow
- water resources
- forest products
- agricultural products
- freshwater products
- saltwater products
- flora and fauna
- wind power
- ocean tidal power
- solar power

↔

Social Welfare

Demographic
- migration
- morbidity
- fertility
- mortality

Economic
- property value changes
- inflation
- opportunity costs
- sectoral effects

Social
- government relations
- regulations
- indigenous people's rights
- quality of life

Community
- public services
- infrastructure
- satisfaction

Family
- structural changes
- stability changes
- employment

Figure 3-6. Possible Impact Categories

3.2 CHARACTERIZATION

Although classification can provide useful information for reviewing the types of impacts associated with specific inventory items, further analysis may be desired to adequately describe those impacts. The characterization phase aims to describe and estimate the contribution of inventory items to environmental impacts via the use of specific impact assessment tools or characterization models. The intended output of the characterization phase is a set of impact descriptors, which include data points or other information that describes the relationship between specific inventory items and impacts.

TABLE 3-1. EXAMPLE INVENTORY ANALYSIS DATA FROM THE MANUFACTURE OF HDPE

Component	Inventory Item	Quantity (mass or volume)
Resource use	Crude oil	....
Energy demand	Electricity	....
	Coal	....
	Renewable fuel	....
	Energy in material	....
Air emissions	CO_2	....
	SO_2	....
	NO_x	....
	CO	....
	Hydrocarbons	....
	Particulates	....
	CFC	....
	Hydrogen	....
Water effluents	Crude oil	....
	Phenol	....
	Nitrogen	....
	Organic carbon	....
	Solid waste	....

Source: Ekvall et al. (1992a).

The complete characterization phase, as defined in this document, includes three separate but complementary activities:

- determining assessment endpoints,
- selecting measurement endpoints (if necessary), and
- applying characterization models to develop impact descriptors.

3.2.1 Determining Assessment Endpoints

After identifying the impacts associated with inventory items and grouping them into impact categories in the classification stage, the practitioner should review the previously

established goals and scope, and other key decision points, of the overall LCA to identify which of the impacts are within the scope of the study, which are referred to as the assessment endpoints. These endpoints represent the focus of the characterization efforts.

TABLE 3-2. EXAMPLE CLASSIFICATION OF INVENTORY ITEMS UNDER IMPACT CATEGORIES FOR HDPE MANUFACTURING

Ecosystem Impacts		Human Health Impacts		Natural Resource Impacts	
Impact Category	**Inventory Item**	**Impact Category**	**Inventory Item**	**Impact Category**	**Inventory Item**
Greenhouse effect	CO_2 CFC Particulates	**Carcinogenic effects**	Crude oil	**Fossil fuel depletion**	Crude oil Fossil fuel
Ozone depletion	CFC	**Lung damage**	Particulates SO_2 NO_x Hydrocarbons	**Renewable energy use**	Renewable fuel
Acidification	SO_2 NO_x	**Odor**	Solid waste Ethylene Oil Phenol		
Smog/fog	NO_x				
Water contamination	Oil Phenol N Organic C Solid waste				
Habitat alteration	Fossil fuel Renewable fuel Solid waste Electricity				
Geomorphic alteration	Electricity Solid waste Fossil fuel Renewable fuel				

Because LCAs are restricted by their goals and scope, many of the impacts identified may not be included in the LCA. The only "correct" set of impacts, or assessment endpoints, is that which satisfies the specific goals and scope of the LCA at hand. Because no single "correct" or minimum set of assessment endpoints should be included in an impact assessment, practitioners should make clear, and possibly qualify, the exclusion of any impacts as assessment endpoints.

For example, SO_2 emissions quantified in the inventory analysis can be associated with acid precipitation, which in turn can lead to a number of further impacts as identified in the classification stage, including the destruction of high-altitude forests, acidification of water bodies, corrosion of buildings and materials, and leaching of metals from soils. The practitioner might determine that, for example, acidification of waterbodies is the most pertinent impact based on the key LCA decision points; thus acidification of waterbodies will be considered an assessment endpoint.

Determining assessment endpoints from a potentially large number and variety of impacts is by no means a straightforward exercise. The key LCA decision points must continually be reviewed, possibly in coordination with defined criteria for guiding the determination of assessment endpoints. Table 3-3 outlines some suggested criteria for determining assessment endpoints. Practitioners should select assessment endpoints that provide useful information for characterizing potential impacts. The assessment endpoints should be selected in an unbiased, scientifically objective manner to help ensure that results of the LCA are unbias and credible.

3.2.2 Selecting Measurement Endpoints

If the assessment endpoint is not directly measurable, then the practitioner may opt to select a measurement endpoint as a surrogate for the assessment endpoint. A measurement endpoint is a measurable characteristic of an impact that can be related to a specific assessment endpoint (EPA, 1992b). When selecting measurement endpoints there may be properties of a specific inventory item, or group of inventory items, for which a surrogate measure (i.e., measurement endpoint) of potential impact can be used. For example, the acid deposition potential of a given amount of SO_2 emissions can be used as a surrogate to link that quantity of SO_2 emissions to impacts such as leaching of metals from soils, tree damage, or fish mortality. If the assessment endpoint is directly measurable, then it can be used as a measurement endpoint.

Because a number of possible measurement endpoints may be available, practitioners need to determine the most appropriate and useful endpoint before beginning the characterization phase of an impact assessment. Using the key LCA decision points as a guide or a set of selection criteria may be helpful when choosing measurement endpoints. Some possible criteria for selecting measurement endpoints that are specific to impact assessment include the following:

- the relevance of the measurement endpoint to the goals and scope of the LCA,

TABLE 3-3. SUGGESTED CRITERIA FOR DETERMINING ASSESSMENT ENDPOINTS

Criteria	Description
Study goals	Good communication between the analyst and the decisionmaker(s) is important to ensure that the chosen assessment endpoints appropriately meet and complement the goals and objectives of the study.
Study scope	Scoping helps to ensure that the goals and objectives of the study are met. The scope of the study defines not only the spatial and temporal boundaries of potential impacts considered but also defines such factors as the intended end use or application of the impact assessment results. If the scope of the study is defined to consider site-specific impacts of deforestation, then site-specific impacts would constitute appropriate assessment endpoints.
Magnitude of environmental loading	The magnitude of environmental loadings as quantified in the inventory analysis could be used to further delimit areas to focus more detailed levels of impact assessment. However, it would be redundant to use the magnitude of environmental loadings as a decision point for more simplistic impact assessment methods (e.g., less is better, relative magnitude).
Environmental relevance	Environmentally relevant assessment endpoints reflect important characteristics of the natural environmental system and are functionally related to other possible endpoints. Changes at higher levels of organization may be of greater significance because of their potential for causing major impacts at lower levels of organization.
Level	The most appropriate assessment endpoint is the earliest impact (i.e., nearest in time to the release of an inventory item to the environment) that allows one to distinguish between alternative impacts or alternative systems. This criterion is most applicable to comparative studies.
Stakeholder values	Stakeholder (including societal) values can range from protection of endangered species to preservation of environmental attributes for functional reasons (e.g., floodwater retention by wetlands) or aesthetic reasons (e.g., visibility in the Grand Canyon).
Data availability	Data availability is a limiting factor that cuts across all fields of research. In some cases, data may be more readily available for one assessment endpoint than another, thus making it a more attractive candidate. However, the convenience of readily available data should not be in lieu of quality. The quality of the available data should be evaluated against previously developed data quality goals.

Source: EPA, 1992d.

- the consistency of an endpoint with the scope and boundaries of the inventory analysis,
- the intended application or end use of the impact assessment results,
- data limitations,
- the availability of impact assessment models, and
- the ease of characterizing potential impacts (i.e., direct versus indirect impacts).

Ideally, the characterization phase will quantify the relationship between an inventory item and an assessment endpoint. When an assessment endpoint can be directly measured, this process can be relatively straightforward. When it cannot be measured, the practitioner must establish the relationship between the inventory item and a chosen measurement endpoint. The practitioner might also use additional extrapolations, analyses, and assumptions to predict or infer changes in the assessment endpoint. It is critical to make these methods and assumptions clear in the final impact assessment results.

3.2.3 Applying Characterization Models to Develop Impact Descriptors

The ability to characterize measurement endpoints hinges on the availability and use of specific impact assessment tools, called characterization models, to describe the contribution of specific inventory items to impacts. The preliminary framework for this characterization activity is contained in a five-tiered hierarchy of characterization models, as described in Table 3-4. This five-tiered hierarchy is based on discussions from the February 1992 SETAC Life-Cycle Impact Analysis Workshop (see SETAC, 1993) and the October 1992 SETAC Life-Cycle Data Quality Workshop.

A primary concern of this characterization activity is the lack of available data for conducting many levels of assessment. At present, data requirements generally increase and data availability generally decreases moving from Tier 1- to Tier 5-type assessments. A recent SETAC-sponsored LCA Data Quality Workshop in Wintergreen, Virginia, recognized that currently available environmental input and output data can only support some Tier 2- to Tier 3-type models. As shown in Table 3-4, advancing to Tier 2- and 3-type assessments requires equivalency factors and chemical-properties (i.e., toxicity, persistence, and bioaccumulation) data. Proceeding to Tier 4- and Tier 5-type models requires high quality, process-specific, unaggregated, and unaveraged inventory data. Data produced from the inventory analysis are currently unable to support most Tier 4- and Tier 5-type assessments. Developing a publicly available database specifically designed for use in LCA to serve as a

clearinghouse for generic information supporting LCAs as well as for other types of residuals-based analyses is a high-priority item within the LCA community.

TABLE 3-4. CHARACTERIZATION MODELS: TIERS OF COMPLEXITY AND ASSOCIATED DATA NEEDS

Tier	Description	Data Needs
Tier 1: Loading Assessment	Inventory data alone are used to evaluate on the basis of quantity or volume with the assumption that "less is better."	Mass, volume, or other units of physical quantity of inventory items.
Tier 2: Equivalency Assessment	Algorithms based on hazard information are used to derive impact equivalency units to evaluate inventory items within a specific impact category.	Same as Tier 1, plus algorithms for equivalency conversions. Also can include resource stock and yield, and non-chemical loading information.
Tier 3: Toxicity, Persistence, and Bioaccumulation Assessment	Interactive properties between a chemical and an organism (toxicity) and an ecosystem (persistence and bioaccumulation) are used to evaluate inventory items.	Same as Tier 1, plus information on characteristics of chemical interactions with organisms (toxicity) and ecosystems (persistence, bioaccumulation).
Tier 4: Generic Exposure/Effects Assessment	Generic environmental or human health information are used to estimate potential impacts of inventory items.	Same as Tier 1, plus generic environmental information and regional calibration model.
Tier 5: Site-Specific Exposure/Effects Assessment	Site-specific environmental or human health information is used to estimate potential impacts of inventory items.	Same as Tier 1, plus site-specific environmental information and a site-specific calibration model.

Source: SETAC, 1993

Loading Assessment

Loading assessment is based on the premise of "less is better" and is the simplest type of characterization method. In loading assessment, the data generated in the inventory analysis are directly used to identify areas where impacts can be reduced through reductions in inputs and outputs. Loading assessment does not assess—qualitatively or quantitatively—the impacts of those inputs and outputs or the benefits of their reduction.

Equivalency Assessment

Equivalency assessment includes approaches that translate inventory items into common units (via the use of equivalency factors) of impact that can either be evaluated to compare the individual contributions of inventory items to impacts or resulting equivalency units to assess the collective contribution of items to impacts. Equivalency factors are based on mechanisms of impact that relate groups of inventory items to specific impacts. Equivalency units can be aggregated within impact categories to provide an estimate of the total level of impact. This method essentially consists of multiplying the values for groups of inventory items (e.g., greenhouse gases) by the appropriate equivalency factors, thus expressing the inventory items in equivalency units (e.g., global warming potential).

Toxicity, Persistence, and Bioaccumulation Assessment

Toxicity, persistence, and bioaccumulation assessment includes those approaches that are more comprehensive than the Tier 2 equivalency assessment approaches because they take into account not only hazard but also ecosystem and organism exposure information. Specifically, these models often focus on properties such as toxicity as an indicator of hazard and persistence and bioaccumulation as indicators of exposure. The main premise of these models is to use information on the inherent properties of substances to assess the potential impacts of chemical substances on the environment.

Information on the inherent properties of many chemical substances can be found in the literature (e.g., environmental fate of organic chemicals or fate-and-transport literature). It can also be predicted using computer databases (e.g., Aquatic Toxicity Information Retrieval, AQUIRE, for water) and models (e.g., Regional Acidification Information and Simulation, RAINS, for acid precipitation).

Generic Exposure/Effects Assessment

Generic exposure/effects assessment is the next higher level of complexity that includes approaches that use generic environmental and human health information to model the potential impacts of inventory items on a generic level. These generic approaches typically utilize computer-based models to determine the fate, transport, and partitioning of substances released to hypothetical, computer-generated "environments." The computer-generated environments contain standardized information on the main components of the environment (i.e., atmosphere, hydrosphere, soil, and biota [plants, animals, and microorganisms]).

Site-Specific Exposure/Effects Assessment

Site-specific exposure/effects assessment approaches utilize general to site-specific environmental and human health information to provide site-specific information on potential impacts. It should be noted that use of detailed, site-specific information should only be needed in cases where such information is required to clarify the decision to be made. The necessary time and resource expense of conducting site-specific studies, as well as data availability limits, makes their applicability to most impact assessments questionable in most cases.

The use of site-specific approaches may be appropriate for some LCAs. However, for many LCAs, site-specific approaches may not be necessary or desirable.

Central to the characterization phase is choosing the characterization model that is the appropriate level of detail to complement the key LCA decision points. The objective at this phase is to match the available data and resources with the minimum level of detail needed to distinguish between alternative impacts or systems. Using models that provide more detailed information is only beneficial if the extra effort provides useful information for decisionmaking. If data or resources are not available to conduct an assessment of the desired level of detail, then a less detailed model can be used if it provides useful information. If the less detailed tool or model does not provide useful information, then the characterization might not be worthwhile.

Figure 3-7 illustrates the decision process through which practitioners choose the characterization model of appropriate level of detail.

3.2.4 Impact Descriptors

The application of characterization models provides an initial description of impact, called impact descriptors. When the characterized impact is both the measurement and assessment endpoint, the practitioner may be able to proceed to the valuation phase of impact assessment relatively easily, provided the practitioner derived the appropriate information for satisfying the key LCA decision points. If the measurement endpoint is used as a surrogate measure for the assessment endpoint, the practitioner may need to relate that measurement endpoint to the assessment endpoint in some manner. One problem with relating measurement to assessment endpoints is that the specific type of output produced is not yet clear, because many models have not been applied in the context of LCA.

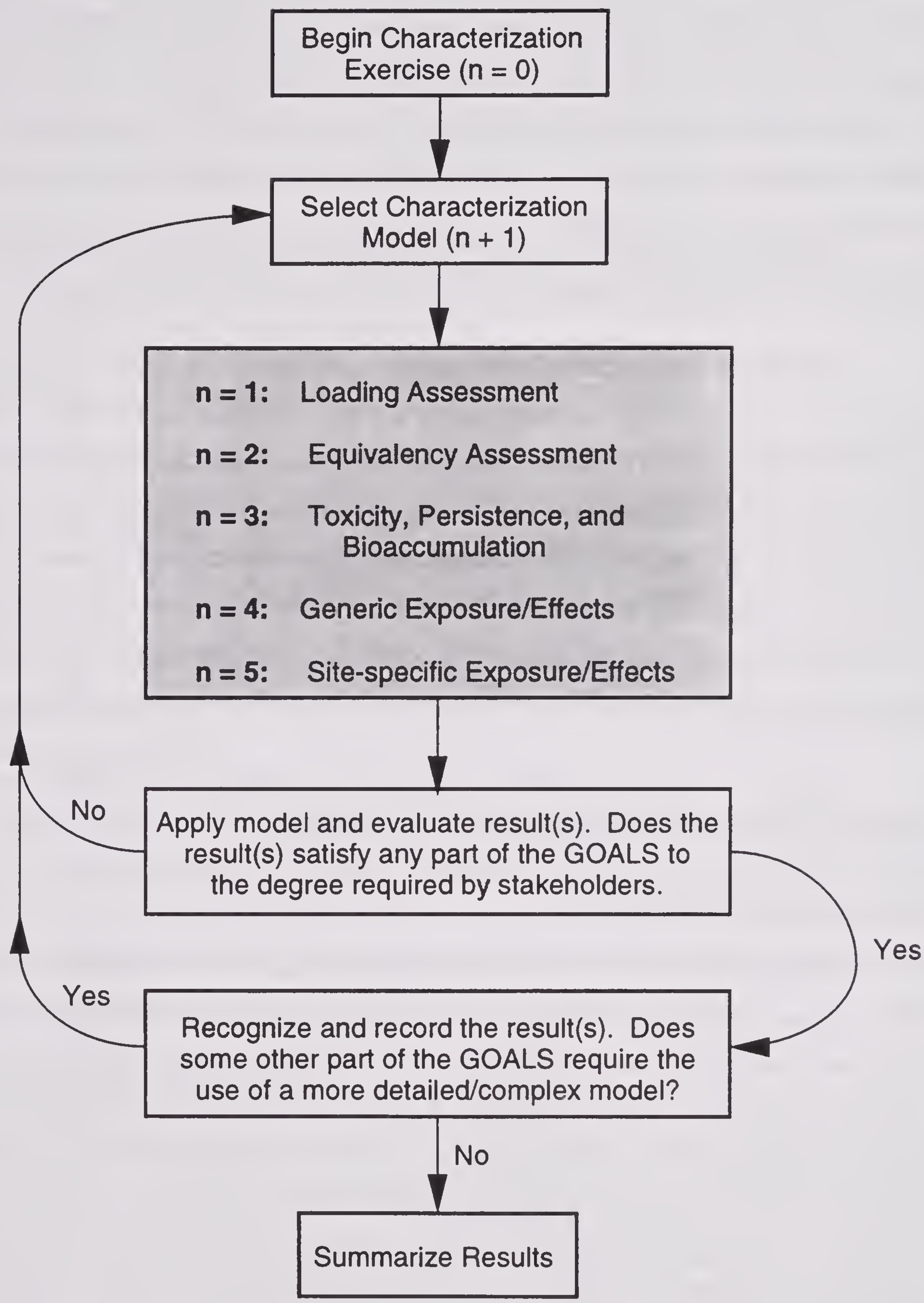


Figure 3-7. Exercise for Choosing Characterization Models

The application of characterization models in impact assessment achieves some aggregation of the inventory analysis and impact characterizations stages of LCA, resulting in a simpler set of impact descriptors within each impact category (SETAC, 1993). Impact descriptors can include quantitative (e.g., numerical level of increase in local tropospheric ozone buildup) and/or qualitative (e.g., descriptive estimate [high, medium, low] of threats to regional wildlife populations) information. In other words, impact descriptors can quantitatively or qualitatively characterize the relationship between specific inventory items and specific impact categories.

3.3 VALUATION

Once a set of impact descriptors has been developed that as concisely and technically possible characterizes the relevant environmental impacts being assessed, the explicit application of valuation methods is appropriate (SETAC, 1993). The valuation phase essentially involves assigning relative values or weights to impacts based on the integration of stakeholder values and the associated impact descriptors.

The main objective of valuation is to establish the relative importance (based on stakeholder values) of multiple impacts to aid in the LCA user's decisionmaking process. Therefore, the practitioner's primary task is to adequately capture and express to decisionmakers the full range of potential impacts relevant to the LCA, without overwhelming his/her audience with information. The practitioner should express these impacts so that determining critical impact areas on which to focus further research and/or improvement efforts is understood.

Although widely practiced, implicitly and explicitly, in the LCA community, the valuation stage is the least developed of the three impact assessment stages. In general, valuation includes the following activities:

- identifying the underlying values of stakeholders,
- determining weights to place on impacts, and
- applying weights to impact descriptors.

Making successful decisions based on impact assessment requires considering all assessment results and technical information. In addition, decisions are not solely based on the precision of measurement but also on how measurements are interpreted in terms of imprecisely understood study goals and stakeholder values. Although developing a truly objective method for valuation may be both impossible and inappropriate, several conceptual and methodological approaches to valuation have been developed (see Chapter 6).

CHAPTER 4

EXISTING METHODS FOR CHARACTERIZING IMPACTS

This chapter provides descriptions of various types of methods for characterizing impacts that have been discussed, presented, or used in the context of LCA. The methods described in this chapter include those that focus on impacts to ecosystems, human health, and/or natural resources. Methods that have specifically been designed to assess resource depletion have historically been kept separate from those methods to assess environmental impacts and are described in Chapter 5. Integrative impact assessment methods that contain a combination of classification, characterization, and/or valuation activities are presented in Chapter 7. Methods evaluated for this document that exhibit potential applicability to impact assessment but which have not been discussed, presented, or used in an LCA context are described in Appendix B.

Although the methods included in this chapter span the three main impact categories (i.e., ecosystem, human health, and natural resources) used in impact assessment, some of the methods are clearly more appropriate for assessing specific impact categories and will be identified as such. Also, because some methods in this chapter do not fit nicely in the generally established five-tier hierarchy of detail for impact characterization, as discussed in Chapter 3, they have not been grouped and/or presented by tier of analysis. The methods however, are presented in the order of increasing level of detail (i.e., from Tier 1 to Tier 5). Table 4-1 provides summary information on each of the methods profiled in this chapter.

4.1 CHECKLIST APPROACH

Inventory analysis provides a quantified listing of inputs and outputs at various stages of the life cycle for a defined product system. The data generated in inventory analysis typically are provided for the weight or volume of input or output (per unit of production or time) either by life-cycle stage or by total for the entire life cycle. Such data alone can be used directly to identify stages in the life cycle where outputs can be decreased. However, the checklist approach merely compares the data generated in the inventory analysis and does not measure impacts. More detailed levels of impact assessment may be required to distinguish the relative environmental importance of various inventory items. Another use of loading data is to compare the overall output levels between alternative products or production systems.

TABLE 4-1. SUMMARY OF METHODS TO CHARACTERIZE IMPACTS

	Impact Categories Covered			Tier of Detail[a]				
Method	**Ecosystem**	**Human Health**	**Natural Resources**	**1**	**2**	**3**	**4**	**5**
Checklist	•	•	•	•				
Relative Magnitude	•	•	•	•				
Environmental Standards Relation	•	•	•		•			
Impact Potentials	•	•	•		•			
Critical Volume	•	•			•			
Environmental Priority Strategy	•	•	•		•			
Tellus Ranking	•	•			•			
TPBP	•	•				•		
Unit World	•						•	
Canonical Environment	•						•	
Ecological Risk Assessment	•							•
Human Health Risk Assessment		•						•

[a]NOTE: Methods are not necessarily confined to any single tier of detail.

The checklist approach is basically a classification matrix that can be used to correlate specific inventory items with specific impacts or impact categories. The checklist allows for the information developed in the inventory analysis to be organized in a meaningful way to provide a quick overview of qualitative impact information. As shown in Table 4-2, the checklist is arranged so that the presence or absence of specific impacts can be clearly shown.

Strengths

The main strength of the checklist approach is its simplicity. Inventory data alone can be used directly without modification, and a simplified view of cause/effect relationships is provided by qualitatively associating impacts and inputs and outputs.

TABLE 4-2. EXAMPLE CHECKLIST FOR ECOSYSTEM IMPACTS

	Inventory Items						
Ecosystem Impacts	**CO_2**	**SO_2**	**O_3**	**CFC**	**Solid Waste**	**Oil Effluent**	**Crude Oil Use**
Atmosphere							
Toxicity			✓				
Ozone depletion							
Greenhouse effect	✓		✓				
Visibility		✓					
Smog/fog			✓				
Ground ozone			✓				
Climate change							
Water							
Toxicity					✓		
Contamination				✓	✓	✓	
Depletion							✓
Thermal							
Turbidity							
Acidification		✓					
Nutrification				✓			
Eutrophication				✓			
Chemical change		✓		✓	✓		
Soil							
Toxicity		✓			✓		
Salinity							
Laterization							
Podzolization							
Erosion							
Other							
Geomorphic					✓	✓	✓
Biodiversity		✓					
Habitat alteration					✓		

In addition to convenience and ease, other strengths of the checklist approach include the following:

- identifying areas for reducing environmental inputs and/or outputs, and
- comparing levels of inputs and/or outputs between alternative materials, processes, or products.

Weaknesses

Although simplicity is the chief strength of the checklist approach, it is also its main weakness. It is critical to recognize that the checklist approach does not actually assess the occurrence of potential impacts or their relative magnitudes.

Some additional weaknesses of the checklist approach include the following:

- choices for environmental improvement are difficult to justify or defend scientifically,
- improvements in environmental conditions may not be achieved because potential impacts are not assessed,
- resources may be wasted on improvement actions that were not part of the real environmental issues, and
- opportunities for environmental improvements may have been missed (SETAC, 1993).

Relevance to Impact Assessment

The checklist approach alone can be used to identify stages in the life cycle where outputs can be decreased. The checklist provides a tool to evaluate the data generated in the inventory analysis. However, the checklist approach does not measure impacts. More detailed levels of impact assessment may be required to distinguish the relative environmental importance of various inventory items. Use of the checklist approach would be more appropriate for internal applications until guidelines are established for the external use of such techniques.

Another use of the checklist approach is to quickly and easily compare the overall input and output levels between alternative products or production systems. Such "quick and dirty" comparisons may not only help identify some key differences between alternatives but also help pinpoint areas to focus more detailed level of analyses.

4.2 RELATIVE MAGNITUDE APPROACH

The relative magnitude approach is another form of loading assessment in which the input and output data generated in the inventory analysis are associated with specific impact categories. Within the specific impact categories, inventory items are further grouped into

subranges based on the level (quantity or volume) of inputs or outputs, thus indicating the relative contribution of various inventory items to specific impacts.

When using the relative magnitude approach, the assigned subrange values may be either subjectively or objectively based. Subjectively based subranges could use a scoring range of 1 to 10 for example, where the inventory items with the lowest quantity would receive a score of 1, and the inventory items with the highest quantity would receive a score of 10. Quantities in between these two bounds can then be extrapolated. Objectively based subranges would use data from the inventory analysis directly (i.e., the actual quantities) as subrange values.

A hypothetical illustration of the type of output derived from the use of the relative magnitude approach is shown in Table 4-3. Although Table 4-3 focuses only on impacts to ecosystems, it can also be used to assess impacts to human health and natural resources.

TABLE 4-3 HYPOTHETICAL EXAMPLE OF THE RELATIVE MAGNITUDE APPROACH FOR ECOSYSTEM IMPACTS

Ecosystem Impact Category	Inventory Item	Quantity (tons)	Subrange Score
Greenhouse Effect	CO_2	4.000	10
	CH_4	0.403	2
	N_2O	0.173	2
	O_3	0.009	1
	CFC	0.001	1
Acidification	SO_2	1.380	10
	NO	0.470	4
	NO_2	0.053	1
Habitat Alteration	Timber	6.000	10
	Coal	5.500	9
	Iron Ore	0.950	1

Strengths

The relative magnitude approach is relatively easy to use; it is based on cause/effect linkages and takes into account the relative quantities of inventory items. In addition, the relative magnitude approach can be helpful for

- identifying areas for reducing environmental inputs and/or outputs, and
- comparing levels of inputs and/or outputs between alternative materials, processes, or products.

Weaknesses

The primary drawback of the relative magnitude approach is its limited capability for comparing different subranges. In addition, as with many other loading assessment approaches, the significance of impacts may be misrepresented because impacts are not measured directly.

Some additional weaknesses of the relative magnitude approach include the following:

- choices for environmental improvement are difficult to justify or defend scientifically,
- improvements in environmental conditions may not be achieved because potential impacts are not assessed,
- resources may be wasted on improvement actions that were not part of the real environmental issues, and
- opportunities for environmental improvements may have been missed (SETAC, 1993).

Relevance to Impact Assessment

The relative magnitude approach can provide a useful screening tool in impact assessment to quickly evaluate the inventory items and impacts that are most significant to the LCA. It may prove particularly useful for screening large numbers of inventory items and impacts. Similar to the checklist approach, however, the relative magnitude approach merely is a tool to evaluate data generated in the inventory analysis and does not provide measures of impact. Thus it is more appropriate to use this approach for internal rather than external application or possibly as a screening tool to pinpoint areas where a more detailed level of analysis is needed.

4.3 ENVIRONMENTAL STANDARDS RELATION (ESR)

The ESR method is a weighting scheme originally developed by Schaltegger and Sturm (1993) to evaluate the environmental impacts of chemical releases in Switzerland. The purpose

of ESR is to assess chemical releases to air, land, and water based on their relative potential ecological and human impact. The information produced from applying ESR can be used to evaluate and compare the relative environmental impacts of alternative products and process or alternative industries. ESR can also be applied to a single process, a collection of processes, or entire systems. Although the use of ESR provides a consistent estimate of the environmental impacts, it does not necessarily preclude the need for additional analyses.

The approach for developing the weights used in ESR is shown in Table 4-4. First, the approach identifies ambient standards (i.e., target concentrations) established by regulatory agencies for chemical levels in air, land, and water that are meant to protect ecosystems and human health. Second, the relationships between the standards were made explicit by converting the ambient standard concentration for each substance in each medium into milligrams per mole. This results in substance- and media-specific standards that are directly comparable. The final step consists of identifying the largest value in all media (in this case substance B, water standard) and then dividing that value by all other values to derive the individual weighting factors. This results in substance- and media-specific weighting factors that are relative to every other chemical in each medium.

The weighting factors have the dimension of pollution units per kilogram (PU/kg) of substance. The environmental impact of a specific chemical release is thus calculated by multiplying the quantity of the released substance by its associated substance- and media-specific weighting factor. The equation for calculating pollution units is as follows:

$$\text{Pollution Units (PU)} = \text{Chemical Emission} \times \text{Chemical- and Media-Specific Weighting Factors}$$

Developing an ESR weighting scheme for the U.S. will not be as straightforward as it was for Switzerland because regulatory standards are much more complex in the U.S. Ideally, regulatory defined and objectively tested ambient standards would be available for all chemical releases of interest to all environmental media. However, this ideal situation does not exist. First, many possible regulatory standards are available in the U.S. for air and water. Second, many of these standards are available for only a few chemicals under one regulatory framework (e.g., National Ambient Air Quality Standards, NAAQS, applies to only six chemicals).

Therefore establishing a decision rule for prioritizing the standards that should be used in the weighting scheme is necessary.

TABLE 4-4. EXAMPLE APPROACH FOR DEVELOPING ENVIRONMENTAL STANDARDS RELATION WEIGHTS

Substance	Ambient Air Standard (mg/m^3)	Ambient Air Standard Expressed in (mg/mole)	Weighting Factor for Air Emissions Pollution Units (PU/kg)
A	1	0.024	30.0
B	12	0.28	2.6
C	4	0.095	7.6
	Ambient Land Standard (mg/kg)	**Ambient Land Standard Expressed in (mg/mole)**	**Weighting Factor for Land Emissions Pollution Units (PU/kg)**
A	5	0.28	2.6
B	10	0.57	1.3
C	8	0.45	1.6
	Ambient Water Standard (mg/l)	**Ambient Water Standard Expressed in (mg/mole)**	**Weighting Factor for Water Emissions Pollution Units (PU/kg)**
A	2	0.036	20
B	4	0.72	1
C	3	0.054	13

Source: Grimstead et al., 1993.

One component of the decision rule is to attempt to develop a weighting scheme using standards that were developed using a consistent approach that is protective of ecosystems and human health and welfare so that the weighting factors for each medium are comparable. For example, if air standards are designed to protect the atmosphere and human health, but water standards are designed only to protect aquatic organisms, then the comparison of pollution units for each medium is less meaningful. It is possible that the water standards would not protect human health; therefore, the water factors would underestimate the potential impacts.

Strengths

Some of the advantages of the ESR weighting scheme include the following:

- ambient regulatory standards represent social, political, regulatory, and scientific opinions and values;
- weighting factors used in ESR consider human health and ecological welfare;
- weighting factors can be derived for all substances that have ambient regulatory standards and/or regulatory values;
- ESR weighting scheme represents the relative impacts of different chemical releases to different environmental media; and
- ESR approach is flexible and can incorporate state, regional, and local regulations for location specific assessments.

Weaknesses

Scientific information on toxicity and environmental health effects are generally considered in establishing ambient standards. However, the ESR weighting scheme's use of relations between ambient standards is not a thoroughly scientific or ecotoxicological-based scheme but instead represents a socio-cultural judgment from an ecological perspective (which relies on ecotoxicological data). No completely objective and undoubtedly valid opinion on the harmfulness of substances exists because of uncertainties in data. Weights for specific pollutants are developed in the ESR method according to generally accepted norms and values, which are theoretically expressed in ambient concentration standards. Such ambient standards may or may not reflect actual environmental impacts.

In addition, the ESR weighting scheme only considers chemical releases. There is no way to account for the environmental impacts resulting from raw materials use, energy consumption, and nonchemical stresses (e.g., noise, heat). It is also critical to recognize that the number of pollution units derived in the ESR weighting scheme represents only one dimension of the overall environmental impact, namely those resulting from pollutant releases. For example, the alteration of pristine habitat, the erosion of fertile top soil, and similar impacts represent a devaluation of environmental assets that is not captured by the pollution units.

Relevance to Impact Assessment

The information produced from applying the ESR can be used in impact assessment to evaluate and compare the relative environmental impacts of inventory items where regulatory

standards exist. Preliminary work is being performed to develop pollutant weighting factors based on U.S. regulatory standards. The use of guidelines and reference concentrations, which are not regulatory standards, may also prove to be useful for the type of analysis. Especially if these reference levels are more strongly based on health and environmental effects rather than technical or economic concerns. For example, EPA inhalation reference concentrations (RfCs) and Maximum Contaminant Level Goals (MCLGs). Although the use of ESR provides a consistent estimate of the environmental impacts, it does not necessarily preclude the need for additional analyses.

4.4 IMPACT POTENTIALS

For some categories of impacts, it is currently feasible to use algorithms to estimate the impact potential of various inventory items. These impact potential algorithms provide a means of converting different types of data generated in the inventory analysis into a common unit for comparison and/or aggregation within impact categories. For example, algorithms for normalizing the contribution of substances to impact categories such as the greenhouse effect have been developed to yield the global warming potential of various substances. Aggregating theses global warming potentials yields a sum figure that can then be used to assess the collective contribution of greenhouse gases to global warming or the contribution of individual greenhouse gases to global warming.

The formula shown below illustrates the generic method for deriving impact equivalency units:

$$\text{Inventory Data} \times \text{Equivalency Factor} = \text{Impact Potential}$$

The inventory data are multiplied by an equivalency factor to yield an impact potential value. Once calculated, the impact potential values can be aggregated within their respective impact categories to assess their collective contribution to the impact category or they can be assessed individually. Table 4-5 describes the state-of-the-art impact potential functions that are available for characterizing specific impact categories.

A major concern of impact potentials is developing equivalency factors for all impact categories that relate inventory data to specific impacts. While it is generally agreed that equivalency factors should be based on impact mechanisms directly related to the impact

categories, it is unclear at this time how equivalency factors would be developed for all categories of impacts.

Impact Potential Example: Ozone Depletion Potential (ODP)

Halocarbons, in addition to being a greenhouse gas, also destroy the stratospheric ozone layer that protects all life from harmful ultraviolet radiation (Graedel and Crutzen, 1990). An ozone hole, amounting to a 50 percent reduction in ozone concentration, now appears over the South Pole in the winter months of the northern hemisphere. Although some features of the Antarctic ozone hole are not fully understood, there is considerable evidence that CfCs are a major cause (Graedel and Crutzen, 1990).

Ozone depletion typically is considered in impact assessment and is included as a major impact category in this document. One way to evaluate ozone depletion in the context of life-cycle impact assessment is to use equivalency units. In this case, ODP units will be used. Table 4-6 shows the type of output from using the ODP algorithm for various halocarbons relative to CFC-11.

Strengths

The primary strength of the impact potentials is that they provide a means of normalizing the contribution of various substances within specific impact categories. This allows for a direct comparison of inventory items to determine which inventory items, or groups of inventory items, contribute most significantly to a specific impact category.In addition, most of the impact potential algorithms are based on cause-and-effect relationships. Thus unlike the checklist and relative magnitude approaches, the impact potentials indicate an estimated environmental impact rather than represent the data generated in the inventory analysis.

Weaknesses

One of the general weaknesses with the impact potentials is that many are based on a large number of assumptions which makes their scientific credibility questionable. In particular, the functions for human toxicity, terrestrial toxicity, and aquatic ecotoxicity potentials are based on a number of debated assumptions which include many inconsistencies. Refer to SETAC (1994) for a complete discussion of the problems and issues related to these three impact potentials.

TABLE 4-5. STATE-OF-THE-ART IMPACT POTENTIAL FUNCTIONS

Impact Potential Function	Description[a]
$GWP_i = \dfrac{\int_0^T a_i c_i(t)dt}{\int_0^T a_{co2} c_{co2}(t)dt}$	The global warming potential (GWP) of a gas is the time-integrated commitment to radiative forcing from the instantaneous release of 1 kg of a trace gas expressed relative to the radiative forcing of 1 kg of carbon dioxide (CO_2): where a_i is the instantaneous radiative forcing due to a unit increase in the concentration of trace gas I, $c_i(t)$ is the concentration of the trace gas I at time t after its release, and T is the number of years over which the calculation is performed.
$ODP(x) = \dfrac{O_3(x)}{O_3(CFC-11)}$	The ozone depletion potential (ODP) is defined as the steady-state ozone reduction calculated for each unit of mass of a gas emitted per year (as a continuous release) into the atmosphere relative to that for a unit mass emission of CFC-11: where ODP(x) is the ODP-value of substance x, $O_3(x)$ is the change in total ozone at steady-state per unit mass emission rate of substance x, and $O_3(CFC\text{-}11)$ is the change in total ozone at steady-state per unit mass emission rate of CFC-11.
$HTP_{i,comp} = \dfrac{DR_i/DF_{icomp}}{DR_{ref}/DF_{refcomp}}$	The Human Toxicity Potential (HTP) is defined as the risk due to an emission flux of 1 kg•year^{-1} of substance I relative to the risk due to an emission flux of 1 kg•year^{-1} of a reference substance: where $HTP_{i,comp}$ is the HTP-value for substance I initially emitted to compartment *comp*, DR_i is the change in human risk at a change of emission flux $DF_{i\ comp}$ (= Dm_i / Dt) of substance I to compartment *comp*, and DR_{ref} is the change in human risk at a change of emission flux $DF_{ref\ comp}$ (= Dm_{ref} / Dt) are the same quantities for the reference substance *ref*.
$TETP_{i,comp} = \dfrac{DR_i/DF_{icomp}}{DR_{phenol}/DF_{phenol\ air}}$	The Terrestrial Ecotoxicity Potential (TETP) is the risk to terrestrial ecosystems (R_i) through an emission-flux of 1 kg of substance I to compartment comp ($DF_{phenol\ air}$ = Dm/Dt) relative to the risk to terrestrial ecosystems (R_{phenol}) through an emission-flux of 1 kg phenol to air ($DF_{phenol\ air} = Dm_{phenol}/Dt$).
$AETP_{i,comp} = \dfrac{DR_i/DF_{icomp}}{DR_{phenol}/DF_{phenol\ air}}$	The Aquatic Ecotoxicity Potential (AETP) is the risk to aquatic ecosystems (R_i) through an emission-flux of 1 kg of substance I to compartment comp ($DF_{phenol\ air}$ = Dm/Dt) relative to the risk to aquatic ecosystems (R_{phenol}) through an emission-flux of 1 kg phenol to air ($DF_{phenol\ air} = Dm_{phenol}/Dt$).

(continued)

TABLE 4-5. STATE-OF-THE-ART IMPACT POTENTIAL FUNCTIONS (CONTINUED)

Impact Potential Function	Description[a]
$POCP = \frac{a/b}{c/d}$	The photochemical ozone creation potential (POCP) is the change in photochemical oxidant production due to a change in emission of the particular volatile organic compound (VOC) relative to the change in photochemical oxidant production due to a change in emission of ethylene: where a is the change in photochemical oxidant formation due to a change in a VOC emission, b is the integrated VOC emission up to that time, c is the change in photochemical oxidant formation due to a change in ethylene emission, and d is the integrated ethylene emission up to that time.
$AP_i = \frac{\text{potential } H^+_i/m_i}{\text{potential } H^+_{SO_2}/m_{SO_2}}$	The acidification potential (AP) is defined as the number of potential H^+ equivalents (H^+_I) per mass unit of substance I (m_i) compared to the number of potential H^+ equivalents (H^+_{ref}) per mass unit of reference substance (m_{ref}); SO_2 is the proposed reference gas.
$NP_i = \frac{N_{equivalents\,i}/m_i}{N_{equivalents\,PO_4}/m_{PO_4}}$	The nutrification potential (NP) is the potential biomass in terms of N-equivalents per unit mass emitted of substance I (m_i) relative to the potential biomass in terms of N-equivalents per mass emitted of a reference substance (m_{ref}); PO_4 is the proposed reference substance.

[a]Many of the functions list in the table are based on a large number of assumptions that are not discussed here.

Source: Guinee and Heijungs, 1993; Guinee, 1992a; and Guinee, 1992b.

Another general weakness with the impact potentials is that only a handful of impact categories (i.e., those listed in Table 4-5) can currently be accounted for with this method. In addition, impact potentials may only be useful for chemical-based inventory items, and not all chemicals are amenable to the development of impact potentials (such as nutrient and oxygen-demanding chemicals).

In addition, a common set of impact potentials still needs to be developed in order for the approach to be used in impact assessment. The applicability of the impact potentials to impact assessment may also be limited because general environmental features or characteristics vary

according to geographic location. This will lead to variation among equivalency units and diminish the utility of a common database of equivalency units. Also, while the development of equivalency factors is straightforward in principle, frequently exposure and effects information on which equivalency factors could be based is lacking. Finally, the multiple mechanisms involved in environmental processes are difficult to identify, making their incorporation into equivalency factors even more difficult.

TABLE 4-6. OZONE DEPLETION POTENTIAL (ODP) OF SELECT HALOCARBON GASES

Gas	ODP Relative to CFC-11	Gas	ODP Relative to CFC-11
CFC-11	1.00	HCFC-141b	0.11
CFC-12	1.00	HCFC- 142b	0.06
CFC-113	1.07	HCFC-143a	0
CFC-114	0.80	HCFC-152a	0
CFC-115	0.50	Halon-1301[2]	16.00
Carbon Tetrachloride	1.08	H-1211	
HCFC-22	0.06	H-1202	
HC FC-123	0.02	H-2402	
HCFC-124	0.02	H-1201	
HCFC-125	0	H-2401	
HCFC-134a	0	H-23 11	

Source: EPA, 1993a.

Relevance to Impact Assessment

While impact potentials provide a relatively simple means for relating inventory data to impact categories, as well as a means for aggregating the data, the delineation of equivalency factors presents a stumbling block. Currently, equivalency factors are being developed for global warming, ozone depletion, acidification, photochemical ozone, nitrification, and biochemical/chemical oxygen demand (BOD/COD).

Detailed examples of the use of impact potentials for determining the GWP and ODP of various emissions are provided in EPA (1993a).

4.5 CRITICAL VOLUME APPROACH

The critical volume approach is a variation on the impact potential approach that is applicable to ecosystem and human health impacts. The critical volume approach is used to determine the volume of air, water, or soil that is needed to dilute specific substances to a generally estimated toxicity threshold. For example, if it was known that the threshold concentration for vegetation was 100 kg of chemical X per 1,000 L of soil volume, and 1,000 kg of chemical X were released, then the critical volume would be 10,000 L of soil.

The results of calculating critical volumes can be grouped into three categories: critical volumes of air, water, and soil. Table 4-7 illustrates an example of applying the critical volume method to a hypothetical set of inventory items.

TABLE 4-7. EXAMPLE OF THE CRITICAL VOLUME APPROACH

Chemical Release	Quantity (kg)	Ecosystem Threshold Levels (kg/L)	Critical Volume (L)
Air			
A	57	.001	57,000
B	88	.001	88,000
C	150	.1	1,500
D	632	.1	6,320
Water			
E	126	.01	12,600
F	17	.001	17,000
Soil			
G	1,000	.1	10,000
H	161	.01	16,100

Strengths

The primary strength of the critical volume approach is that it provides a means of normalizing a variety of data to a common measure (i.e., critical volume in liters) of environmental impact. The critical volume approach is relatively simple and convenient to use

and produces useful results. In addition, similar to the impact potentials described in Section 4.4, the calculations used in the critical volume approach are based on toxicity and exposure concepts, which are already familiar to environmental analysts.

Weaknesses

To efficiently and successfully use the critical volume approach, a new understanding and methodology for the impact equivalency approach must be created. However, exposure and toxicity information is generally lacking for many environmental and human impact areas with which to determine critical volume values.

In addition, the critical volume approach only takes into account the assimilation of one chemical at a time. This is, the approach does not take into account the interaction between multiple chemical releases to the same environmental media. For example, Table 4-7 shows 10,000 L and 16,100 L as the critical volumes of soil needed to dilute 1,000 kg and 161 kg of chemicals G and H, respectively, to generally accepted threshold concentrations. What is not provided is an indication of how these critical volumes might be affected as both chemicals are released to the same medium in the same location.

Relevance to Impact Assessment

The critical volume approach can provide a relatively familiar framework (i.e., exposure and toxicity concepts) for normalizing and comparing largely different types of inventory items. In addition, the critical volume approach can provide a simplified means of normalizing data generated in inventory analysis by expressing them in terms of volumes, which are then amenable to aggregation into common impact categories.

In the context of impact assessment, the critical volume approach would be most useful for characterizing chemical releases. However, critical volume algorithms are currently available for only a limited number of chemicals, and the approach does not lend itself to assessing nonchemical components.

4.6 ENVIRONMENTAL PRIORITY STRATEGY (EPS)

The Federation of Swedish Industries and the Swedish Environmental Research Institute initiated an Environmental Priority Strategies (EPS) system in collaboration with the Volvo Car Corporation. Although based on implicit value judgments regarding the environmental impacts of various substances, the EPS system nonetheless provides a means of calculating, in semi-quantitative terms, the overall environmental impact of a product system.

The EPS system employs environmental indices to convert various material uses and emissions quantified in the inventory analysis into measures of impacts. These indices are calculated by carrying out the following steps: (1) each material use or emission being evaluated is assigned one score for each of the factors listed below, and (2) the six factor scores are then multiplied together to yield a single score. This single score is expressed in a measure called the environmental load unit (ELU).

The factors that are assigned scores to calculate indices are the following:

1. Scope—the general impression of the environmental impact
2. Distribution—the extent of the area affected
3. Frequency and/or Intensity—the regularity and intensity of the problem
4. Durability—the permanence of the effect
5. Contribution—significance of 1 kg to the total impact
6. Remediability—relative cost to reduce the emission

The higher the ELU of a material, the higher its contribution to an impact and vice versa. Table 4-8 presents selected environmental indices for raw materials and energy use and for releases to the air, water, and soil.

Once the indices are determined, the environmental load value (ELV) is determined as a description of the impacts of the material use or emission in question. The ELV is calculated, as shown below, by multiplying the quantity of the material use or emission by its environmental index (typically expressed as ELU per kilogram). Table 4-9 illustrates some generic ELVs using hypothetical inventory analysis data.

$$\text{Environmental Load Value} = \text{Environmental Index (ELU)} \cdot \text{Quantity}$$

Strengths

The primary strength of the EPS system is its flexible framework, which allows analysts to normalize impacts for direct comparison of inventory items either within or between specific impact categories. A number of environmental load indices have been developed to date, thus allowing for a relatively comprehensive assessment of environmental impacts. Using the environmental load indices for specific materials and processes enables the user to calculate ELVs for individual activities, processes, or an entire system.

TABLE 4-8. SELECT ENVIRONMENTAL INDICES USED IN EPS

Index	Measure	Index	Measure	Index	Measure
Raw Materials	*(ELU/kg)*	*Air Emissions*	*(ELU/kg)*	*Water Emissions*	*(ELU/kg)*
Co	12,300	CO_2	0.04	Suspended matter	1E-07
Cr	22.1	CO	0.04	BOD	0.0001
Fe	0.38	No_x	245	COD	0.00001
Mn	21	N_2O	0.6	TOC	0.00001
Mo	4,200	So_x	6.03	Oil	0.00001
Ni	700	CH	10.2	Phenol	1
Pb	363	PAC	600	Phosphorus	2
Pt	42,000,000	Aldehyde	20	Nitrogen	10
Rh	42,000,000	Hcl		DDT	10,000
Sn	4,200	F	1E-07	PCB	10,000
V	42	Hg	10	Dioxin	100
Oil	0.168	Cd		Al	1
Coal	0.1			As	0.01
Land	*(ELU/m^2)*	*Soil Emissions*	*(ELU/kg)*	Cd	10
Arable	2.93	As		Cr	0.5
Forested	1.05	Cd		Cu	0.005
Residual	0.98	Cr		Fe	1E-07
Energy	*(ELU/kg)*	Cu		Hg	10
Oil	0.33	Hg		Mn	1E-07
Coal	0.26	Ni		Ni	0.001
Electricity	0.014	Pb		Pb	0.01
		Sn		Zn	0.00001
		Ti			

Source: Swedish Environmental Research Institute, 1991.

Weaknesses

One of the weaknesses of the EPS systems is in its primary assumption that a linear relationship between ELVs and increasing or decreasing quantities of inventory items exists. In reality, this relationship is probably not linear but more complex. Another weakness of the EPS system is its reliance on value judgments to develop the environmental indices used to calculate

the ELVs. For example, defining the scope or "general impression of the environmental impact" of the indices is subject to different interpretations by different stakeholders or by different geographic locations. Also, EPS does not appear to adequately consider relative toxicity of pollutants. Thus the robustness of the indices is open to debate. The same point could also be made for the distribution and remediability components.

TABLE 4-9. EXAMPLE ENVIRONMENTAL LOAD VALUES

Inventory Item	Quantity (kg)	Environmental Index (ELU)	Environmental Load Value
Raw Materials			
Oil	1,000	0.168	168
Fe	450	0.38	171
Pb	123	363	44,649
Ni	37	700	25,900
Mn	25	21	525
Energy			
Oil	1,500	0.33	495
Electricity	20,000	0.014	289
Air Emissions			
CO_2	390	0.04	15.6
NO_x	375	245	91,875
SO_x	248	6.03	1,495
Water Emissions			
Oil	29	0.00001	0.00029
BOD	58	0.0001	0.0058
Pb	5	0.01	0.05

Source: Swedish Environmental Research Institute, 1991.

Relevance to Impact Assessment

The ELVs developed through the EPS system are normalized measures of environmental concerns that can be used to compare inventory items, or they can be aggregated to compare life-cycle stages or entire product systems. The ELVs can also be used to compare the environmental impact profiles of alternative materials, processes, or products.

The EPS system is currently used in some European LCAs and thus provides a practical methodology for assessing the impacts of inventory items on the environment. However, the EPS system is currently set up primarily to assess impacts to ecosystems. If developed and refined, modules for assessing impacts to human health and natural resources would enhance the overall utility of EPS.

4.7 TELLUS INSTITUTE HUMAN HEALTH HAZARD RANKING

One human health hazard ranking approach, used by Tellus Institute in an assessment of packaging materials (Tellus Institute, 1992a, 1992b, 1992c), groups inventory items into two main categories for assessment: carcinogens and noncarcinogens. This section describes the methods used to assess each of these impact groups.

The Tellus approach assesses the relative human health carcinogenic impact of substances based on a cancer potency factor, which is measured in milligrams/kilogram of body weight/day. The cancer potency factor is designed to represent the cancer risk associated with various inventory items. Isophorone was chosen as a baseline of comparison for the substances, because it possesses the lowest cancer potency. The calculated potency factors for various substances are shown in Table 4-10.

Noncarcinogenic substances were assessed on the basis of each substance's oral reference dose. While reference doses (RfDs) can be determined by two routes of exposure—oral and inhalation—Tellus used oral RfDs because many more oral RfDs are available in the literature. The oral reference dose provides an estimate of the maximum daily level of exposure that will not cause harm and is measured in milligrams substance/kilogram body weight/day (Tellus Institute, 1992b).

The higher the RfD, the less toxic the substance, since a higher dose is needed for an effect to occur. In Tellus's ranking, the inverse of the RfD was used as the ranking factor in order for the ranking number to be indicative of lower toxicity. The baseline substance for the noncarcinogenic ranking was xylene, because it has the highest RfD (i.e., smallest inverse), which indicates xylene is the least toxic of the set of pollutants. Thus the inverse RfDs are compared to xylene to derive "xylene equivalents." Table 4-11 illustrates some of the RfDs and xylene equivalents for specific substances.

TABLE 4-10. CARCINOGEN POTENCY FACTORS AND ISOPHORONE EQUIVALENTS

Substance	Cancer Potency	Isophorone Equivalents
Acrylonitrile	5.40E-01	138
Arsenic	5.00E+01	12,821
Benzene	2.90E-02	7
Beryllium	4.30E+00	1,103
Bis(2-ehtylhexl) phthalate	1.40E-02	4
1,3-Butadiene	1.80E+00	462
Cadmium	6.10E+00	1,564
Carbon tetrachloride	1.30E-01	33
Chloroform	6.10E-03	2
4,4-DDT	9.70E-06	0.00249
1,4-Dichlorobenzene	2.40E-02	6
1,2-Dichloroethane	9.10E-02	23
1,1-Dichloroethylene	6.00E-01	154
1,2-Dichloropropane	6.80E-02	17
1,3-Dichloropropene	1.80E-01	46
2,4-Dinitrotoluene	6.80E-01	174
2,6-Dinitrotoluene	6.80E-01	174
1,2-Dipenylhydrazine	8.00E-01	205
Ethylene oxide	3.50E-01	90
Hexachlorobenzene	1.60E+00	410
Isophorone	3.90E-03	1
Methylene chloride	7.50E-03	2
Nickel	8.40E-01	215
PAHs	1.15E+01	2,949
Propylene	2.40E-01	62
Styrene	3.00E-02	8
Tetrachloroethylene	5.10E-02	13
1,1,1-Trichloroethane	5.70E-02	15
Trichloroethylene	1.10E-02	3
Vinyl Chloride	2.30E+00	590

Source: Tellus Institute, 1992b.

TABLE 4-11. EXAMPLE RfDS FOR NONCARCINOGENIC RANKING

Substance	Reference Dose (oral)	1/RfD	Xylene Equivalents
Acetone	1.00E-01	10	20
Antimony	4.00E-04	2,500	5,000
Arsenic	1.00E-03	1,000	2,000
Barium	5.00E-02	20	40
Beryllium	5.00E-03	200	400
Cadmium	5.00E-04	2,000	4,000
Chromium	1.00E+00	1	2
Copper	3.71E-02	27	54
Cyanide	2.00E-02	50	100
4,4-DDT	5.00E-04	2,000	4,000
Fluoride	6.00E-02	17	33
Hydrogen Sulfide	3.00E-03	333	667
Lead	1.40E-03	714	1,429
Manganese	2.00E-01	5	10
Mercury	3.00E-04	3,333	6,667
Napthalene	4.00E-03	250	500
Nickel	2.00E-02	50	100
Phenol	6.00E-01	2	3
Selenium	3.00E-03	333	667
Tin	6.00E-01	2	3
Toluene	3.00E-01	3	7
Zinc	2.00E-01	5	10

Source: Tellus Institute, 1992b.

Once the carcinogenic and noncarcinogenic rankings have been developed, the analyst may want to determine the relationship between the two groups of substances. To accomplish this, Tellus used the Occupational Safety and Health Administration (OSHA) permissible potency level (PEL) figures. For xylene, the PEL is 100 parts xylene per million parts of air (ppm), and for isophorone, the PEL is 25 ppm. Converting the ppm units into milligrams, the 100 ppm PEL for xylene translates to 433 mg/m^3, and the 25 ppm PEL for isophorone translates into 141 mg/m^3. From these conversions, one might deduce that a "safe" dose of xylene is three

times the "safe" dose of isophorone; thus, isophorone has a xylene equivalent factor of three. This relationship can be used to compare and determine the combined effect of the carcinogenic and noncarcinogenic groups as shown in Table 4-12. In this approach, Tellus used isophorone and xylene equivalents. Other possible equivalent factors for impacts to human health include the following:

- **PELs**—specify the amount of a pollutant to which a worker can be exposed over an 8-hour work day.
- **Threshold Limit Values (TLVs)**—specify the amount of a substance a worker can be exposed to over an 8-hour work day.
- **Short-Term Exposure Limits (STELs)**—only established for a small number of chemicals and may not be useful for assessing potentially large numbers of substances found in a typical life-cycle inventory.
- **Immediately Dangerous to Life and Health (IDLHs)**—only established for a small number of chemicals and may not be useful for assessing potentially large numbers of substances found in a typical life-cycle inventory.
- **Maximum Concentration Levels (MCLs)**—used by the Safe Drinking Water Act to establish regulations for pollutants in public water systems. However, MCLs have only been established for a few substances.

Strengths

The Tellus approach provides a practical example of impact characterization within the context of LCA. This approach assesses the relative human carcinogenic and noncarcinogenic impacts of substances based on cancer potency factors and RfDs, respectively. These techniques take into consideration well-refined and accepted health effects information to estimate relative toxicity. The Tellus approach also provides a means of normalizing and evaluating the relative impact of a variety of different substances.

Weaknesses

The main weakness of the Tellus ranking method is its dependence on a relatively simplistic approach to comparison of cancer to noncancer, and a corresponding lack of transitivity in ranking substances. In addition, for many substances, cancer potency factors and RfDs have not yet been established, and establishing these factors in the near future may not be feasible. In addition to a lack of key information, the Tellus ranking only considers carcinogenic and noncarcinogenic human health impacts. It does not consider persistence or bioaccumulation, and does not look at ecological impacts. For the purposes of impact assessment, it may also be

useful to have a means of considering the specific human health impacts (e.g., mutagenic and teratogenic impacts).

TABLE 4-12. EXAMPLE HUMAN HEALTH IMPACT EQUIVALENCY RANKING

Substance	Carcinogens Isophorone Equivalents	Noncarcinogens Xylene Equivalents	Combined Ranking[a]
Arsenic	12,821	2,000	20,231
Benzene	7		22
Beryllium	1,103	400	1,854
Cadmium	1,564	4,000	4,346
Chloroform	2	200	102
4,4-DDT	2.49E-03	4,000	2,000
Isophorone	1	10	7
Nickel	215	100	373
Styrene	8	10	17
Toluene		7	7
Vinyl Chloride	590		1,769

[a]The combined ranking assumes 1 Isophorone Equivalent = 3 * Xylene Equivalent

$$\text{Combined Rank} = \frac{3\,[(\text{Carcinogenic Equivalents}) + (\text{Noncarcinogenic Equivalents})]}{2}$$

Source: Tellus Institute, 1992b.

Relevance to Impact Assessment

The Tellus approach provides a means for normalizing and comparing both human carcinogenic and noncarcinogenic substances to perform cross-substance comparisons of the potential impact of those substances to human health. In addition, the Tellus ranking methods allows for determining the aggregate contribution of various life-cycle stages or of alternative products and processes to human health impacts.

4.8 TOXICITY, PERSISTENCE, AND BIOACCUMULATION PROFILE (TPBP)

The toxicity, persistence, and bioaccumulation (TPBP) considers the potency (toxicity) as well as the physical and chemical properties of substances to assess their fate and potential environmental impacts (SETAC, 1993). The TPBP can be used in two ways:

- to construct mobility, persistence, and effects profiles for each of the environmental loading factors listed in the inventory analysis, and
- as a screening tool based on identifiable thresholds to determine whether to proceed to a more detailed level of assessment (i.e., Tier 4 or Tier 5).

Input information used in the TPBP can be found in generally accepted testing studies, such as the following, which are readily described in the public and private literature:

- acute toxicity testing (LC50, EC50, TD50);
- chronic toxicity testing (NOEL);
- biodegradation (half life, CO_2 evolution); and
- bioaccumulation (solubility, octanol/water coefficient, bioaccumulation factor) (SETAC, 1993).

When input data for TPBP are not accessible or do not exist, using predictive structure activity relationships from computerized databases may be possible (SETAC, 1993). In addition, for many substances, this information can be predicted using computer models or databases. Table 4-13 provides an example of how this information is used to describe potential environmental impacts.

Strengths

The strengths of the TPBP include the following:

- impacts to ecosystems and human health may be considered;
- input data for the TPBP are generated from generally accepted testing methodologies (e.g., acute and chronic toxicity testing);
- output from the TPBP may be used to identify priority substances for which more detailed levels of analysis may be desired; and
- TPBP, unlike the previously described methods, considers information on toxicity, persistence, and bioaccumulation.

TABLE 4-13. HYPOTHETICAL EXAMPLE OF TPBP APPROACH

Life-Cycle Substance	Life-Cycle Quantity (kg)	Acute Toxicity (LC_{50})	Chronic Toxicity (NOEL)	Biodegradation (half life)	Bioaccumulation Factor
A	50	...	...	...	...
B	32	...	...	...	...
C	246	...	...	...	...
D	65	...	...	...	...
E	97	...	...	...	...
F	32	...	...	...	...
G	5	...	...	...	...
H	43	...	...	...	...
I	785	...	...	...	...
J	324	...	...	...	...
K	17	...	...	...	...

Weaknesses

The primary weaknesses of the TPBP is that it does not consider environmental exposure and can only be applied to a limited number of substances—mainly chemicals (SETAC, 1993). It is not clear how the TPBP would be applied to nonchemical components of ecosystems and/or human health; although it seems possible to identify parameter and thresholds, no attempt has been made to do so (Vigon and Evers, 1992).

In addition, there is no consensus about the indices or measures that are best to use, and it is unclear whether estimated values are of acceptable accuracy for LCA or how this information should be interpreted in the context of impact assessment (Vigon and Evers, 1992). This approach could be expanded upon by considering other health effects data, such as cancer potency values and RfDs, and ecological toxicity information.

Relevance to Impact Assessment

The TPBP provides a further level of detail for impact equivalency type assessments because it considers not only hazard (toxicity) information but also exposure (persistency and

bioaccumulation) information. The desired characteristics of the results obtained from impact assessment may require that such exposure information be considered. In addition, exposure information may be needed to distinguish among ambiguities in impact equivalency results. The properties of TPBP make it a good candidate to use in developing priority listings of substances listed in the inventory that may require assessment at a greater level of detail.

4.9 MACKAY UNIT WORLD MODEL

The Mackay unit world model helps to explain the mechanisms and rates by which toxic substances are transported into and transformed within the natural environment. This approach was originally developed as a means of assessing the likely environmental behavior and effects of newly developed or used chemical compounds before their release into the market and the environment (Mackay, 1979).

The underlying concept of the unit world model is fugacity. Fugacity can be regarded as the "escaping tendency" of a chemical substance from a phase. It has units of pressure and can be related to concentration. Just as temperature (°C) can be related to heat concentrations (cal/m^3) using a proportionality constant, to yield a heat capacity (cal/[m^3 x °C]), fugacities (f) can be related to concentrations using a similar fugacity capacity constant Z, with units of mol/m^3atm by the following equation:

$$C = Zf$$

where Z depends on temperature, pressure, the nature of the substance, and the medium in which it is present. Its concentration dependence is usually slight at high dilution.

The physical significance of Z is that it quantifies the capacity of the phase for fugacity. At a given fugacity, if Z is low, C is low—thus only a small amount of substance is necessary to exert the escaping tendency. Toxic substances thus tend to accumulate in phases where Z is high or where high concentrations can be reached without creating high fugacities.

> **Example:** The fugacity capacity Z for oxygen in water at room temperature is 1.5 mol/m^3 atm (i.e., 0.3/0.2). In air it is 40 mol/m^3 atm (i.e., 8/0.2), a ratio of about 27. Oxygen then adopts a concentration in air 27 times that in water. Conclusion: if we can find Z for a substance for each environmental phase, we can easily calculate how the substance will partition. It will reach highest concentrations where Z is highest (Mackay and Paterson, 1981).

Fugacity is used in the unit world approach to calculate partitioning. The unit world is a hypothetical 1 km^3 box that contains water, soil, air, sediment, and aquatic biota. Mathematically, the unit world is represented by a set of thermodynamic equations that describe the partitioning and transformation of a chemical introduced into the box. Chemical-specific parameters are used to predict the partitioning of a given quantity of chemical among the different components of the unit world. To calculate environmental partitioning in the form of amounts in each medium, it is necessary to assume volumes for each medium and an amount of solute. Medium volumes are based on unit world volumes, consisting of 1 km^2 with a 10 km high atmosphere. In this unit world, 30 percent of the area is covered by soil at a depth of 3 cm, and 70 percent is covered by water at an average depth of 10 cm (with 3 cm of sediment, 5 ppm volume of suspended solids, and 0.5 ppm biota). The corresponding volumes of these five components are as follows:

- Atmosphere—accessible volume $10^{10}m^3$
- Soil—accessible volume 10^5m^3
- Water—accessible volume 10^6m^3
- Sediment—accessible volume 10^4m^3
- Aquatic Biota—in 10^6m^3

The fugacity for each of the compounds is calculated as follows:

- **Pure Substance**
  A pure substance (solid or liquid) has a fugacity that is approximately equal to its vapor pressure (P^S). If its molar volume is v(m^3/g mol), then $Z = C/f = 1/P^S v$. The temperature dependencies of P^S and v (and hence Z) are available in handbooks.
- **Vapor Phase or in the Atmosphere**
  Fugacity is usually equal to partial pressure (P); thus, from the gas law, if n is mols and V is volume, $Z = C/f = n/VP = 1/RT$. In the vapor phase Z is independent of the nature of the substance and is usually about 40 g mol/m^3 atm.
- **Liquid Phase or Water Bodies**
  Fugacity or partial pressure is usually related to concentration by the Henry's Law constant, H as P = PC. It follows that Z is simply 1/H. H is easily calculated as the ratio of pure substance vapor pressure to solubility.
- **Sorbed Phases**
  If the sorption partition coefficient Kp is the ratio of sorbed concentration (g/Mg or ppm) to water concentration (g/m^3 or ppm), and if the sorbent concentration is S(g/m^3), it can be shown that Z is 10^{-6} KpS/H.

- **Biotic Phase**
  If the biota is regarded as part (fraction y) octanol and the volume fraction of biota is B, then Z is By Kow/H, where Kow is the octanol water partition coefficient. Sorbed and biotic phases are the most difficult, but recent work indicates that sorption and bioconcentration can be related to Kow and organic and lipid contents, thus providing good estimates for Z (Mackay, 1979).

By estimating the rates of transformation of the chemical (due to photolysis, oxidation, biodegradation, or other processes), the unit world approach can be used to predict steady-state concentrations, residence times, and removal rates. An example of the type of output information derived from the unit world approach is shown in Figure 4-1. As shown in this figure, the Mackay unit world approach provides the user with information on the partitioning and concentration of a substance in air, water, soil, sediment, suspended solids, and biota. Thus, this approach enables the user to determine and compare the impact of various substances on individual components of the environment. For more detailed explanations of the unit world approach, see Mackay (1979) and Mackay and Paterson (1981).

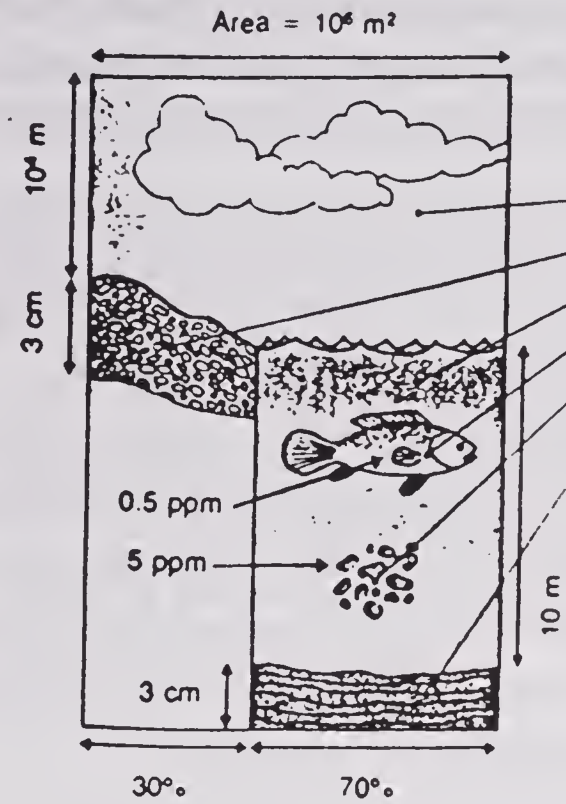
Strengths

The unit world model is a relatively simple approach that has been refined and widely used over the past 15 years to quantify the environmental transformation and fate of chemicals. Input data for the unit world model, which primarily include data on chemical toxicities, exist for many different chemicals and are available on readily accessible databases, such as the EPA's AQUIRE database. However, data on toxicities to terrestrial plants and animals are more scarce (SETAC, 1993).

Weaknesses

Drawbacks of the unit world model are that it focuses only on the fate of chemicals; human health effects are not included. Some additional weaknesses of the unit world model include the following:

- results cannot be validated by experimental observation,
- data are lacking on many chemical substances, and
- currently no practical application exists to serve as a case study example.



	Volume, m^3	Z, mol/($m^3 \cdot$ Pa)	Amount, mol	Concentration mol/m³	Concentration μg/g
Air	10^{10}	4.03×10^{-4}	55.0	5.50×10^{-9}	6.96×10^{-4}
Soil	9×10^3	1.90×10^1	2.33	2.59×10^{-4}	2.59×10^{-2}
Water	7×10^6	0.333	31.8	4.54×10^{-6}	6.82×10^{-4}
Biota	3.5	62.8	0.003	8.56×10^{-4}	1.28×10^{-1}
Suspended solids	35	37.9	0.018	5.17×10^{-4}	5.17×10^{-2}
Sediment	2.1×10^4	37.9	10.9	5.17×10^{-4}	5.17×10^{-2}
			100.0		

Fugacity = 1.363×10^{-5} Pa Temperature = 25 °C

Media densities (kg/m³):

Air = 1.19 Soil = 1500 Water = 1000
Biota = 1000 Suspended solids = 1500 Sediment = 1500

Organic carbon contents:
Soil = 2% Suspended solids = 4%
Sediment = 4%

Figure 4-1. Example Output from the Unit World Model

Source: Mackay, 1979.

Relevance to Impact Assessment

The unit world approach could provide a rational and realistic tool for impact assessment that enables diverse inventory data to be described in terms of environmental medium partitioning, concentration ratios, and overall persistence. This approach may also enable the analyst to determine the sensitivity of each characteristic (e.g., persistence) as a function of the input data, by altering these data and observing the resulting effect.

One potential problem with using the unit world model in LCA applications is that in many cases the user will not know the correct, proportionate amount of chemical released into the unit world box (Vigon and Evers, 1992). For example, if an inventory analysis revealed that a system released 100 grams of NO_x to the air per unit production and this 100 grams was directly inputted into the model, then the equilibrium partitioning would show the relative

concentrations of NO_x in each of the model components. However, without proportionately loading the 100 grams of NO_x into the unit world box, the concentrations produced would be meaningless for comparison against toxicological standards (Vigon and Evers, 1992).

An LCA-specific case study using the unit world model is needed to better assess its relevance to impact assessment.

4.10 CANONICAL ENVIRONMENT MODELING

Canonical environment modeling is similar to, but somewhat more complex and realistic than, the unit world approach. Instead of using a 1 square kilometer unit world, the canonical approach uses a simulated reference environment, or a "canonical environment," such as a generalized stream, lake, pond, or other ecosystem type. Canonical environments do not usually represent any specific real ecosystem; rather, they are representative of a class of ecosystems within a general region.

In contrast to the relatively small number of parameters needed by the unit world approach, canonical environment modeling generally requires a wide variety of environmental parameters (e.g., soil organic matter content, stream flow). Canonical environment models are routinely used by EPA and other organizations for ecological risk assessments (see Barnthouse et al., 1984 and 1985; Suter et al., 1985a and 1985b).

To date, many applications of the canonical environment approach have focused modeling efforts on aquatic systems. However, similar approaches have been established for assessing the fate of pollutants in terrestrial systems. Examples of such approaches are found in Barnthouse et al. (1985a and 1985b) and Suter et al. (1984 and 1985). These models simulate atmospheric dispersion and deposition of pollutants on soil and uptake of pollutants by biota (plants and animals).

EPA's Office of Toxic Substances (OTS) also uses the canonical environment concept to evaluate the fate of pesticides in generic rivers, lakes, and estuaries as part of its Exposure Analysis Modeling System (EXAMS).

Strengths

Canonical environmental modeling provides information on the fate and transformation of chemical releases in various environmental media (e.g., air, water, soil, biota). Such

information enables analysts to consider not only the level of pollutants released to various environmental media (i.e., loading assessment), but also the ultimate fate of those pollutants.

Canonical environmental models also have routinely been used by EPA and other organizations for ecological risk assessments. Although many of these applications have been for modeling aquatic systems, similar approaches have been developed for terrestrial and atmospheric modeling. A wide body of practical examples and experience are available for potential users to draw upon for guidance.

Weaknesses

One weakness of canonical environment modeling is that there are no means to account for nonchemical factors and to directly account for impacts to human health. In addition, it might be (in most cases) uncommon that threshold levels of toxicity, etc., will be exceeded by the environmental releases of any system acting alone in a given region. It is unclear how the canonical models would handle cumulative releases from multiple facilities within a given region. Canonical environment modeling also does not measure impacts per se, but rather the fate and transformation of pollutants in different environmental media. Although such information can provide a useful proxy for "impacts," most environmental components have some level of assimilative capacity, so assuming that the fate of a pollutant in a specific environmental media will necessarily impact that component can be misleading.

Relevance to Impact Assessment

In the context of impact assessment, canonical environment models (at their present state of development) would be most useful for characterizing impacts to ecosystems or resource supplies (e.g., water bodies, forests). Although applications of canonical models to assess impacts to animal populations have been performed, using these models to assess impacts to human health is not clear.

Canonical environment models also may be useful in impact assessment when information (including fate and transformation) on an additional level of detail is needed to distinguish between a number of different pollutants releases to the same or different environmental media.

4.11 ECOLOGICAL RISK ASSESSMENT

Within the last 3 years, two independent groups—the EPA Risk Assessment Forum and the National Academy of Science (NAS) Committee on Risk Assessment Methodology—have

attempted to develop paradigms for ecological risk assessment. EPA defines ecological risk assessment as a process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors. Stressors are defined as any physical, chemical, or biological entity that can induce adverse effects on individuals, populations, communities, or ecosystems (EPA, 1992a and 1992b).

The EPA's current framework for ecological risk assessment is conceptually similar to the risk assessment approach used for human health risk assessment, as outlined in the 1983 NAS report, "Risk Assessment in the Federal Government: Managing the Process." However, ecological risk assessment can be distinguished from human health risk assessment by three primary concepts:

- Ecological risk assessment can consider effects beyond those on individuals of a single species and examine entire populations.
- There is no single set of ecological values to be protected that can be generally applied. Rather, these values are selected from a number of possibilities based on both scientific and policy considerations.
- There is an increasing awareness of the need for ecological risk assessments to consider nonchemical as well as chemical loadings (EPA, 1992a and 1992b).

The EPA conceptual framework for ecological risk assessment is illustrated in Figure 4-2. This framework consists of three major phases:

1. **Problem Formulation:** a planning and scoping process that establishes the goals, breadth, and focus of the risk assessment. The process of problem formulation begins with characterizing ecological exposure to loadings and ecological effects, which includes evaluating loading characteristics, evaluating the ecosystem potentially at risk, and evaluating the expected or observed ecological effects (EPA, 1992a). The output of the problem formulation is a conceptual model that provides a qualitative description of how a given loading can affect an ecological component.
2. **Analysis:** a process of developing profiles of environmental exposure and the effects of stressors that involve two primary activities: characterization of exposure and characterization of ecological effects. The outputs of this phase of the risk assessment are exposure and loading-response profiles that serve as input to the risk characterization phase described below.
3. **Risk Characterization:** a process that integrates the exposure and effects profiles (EPA, 1992a). Risk characterization involves two distinct activities: risk estimation and risk description. The ecological risk summary provides a summary of risk estimation and uncertainty analysis results and assesses the level of confidence in the risk estimates through a discussion of the weight of evidence.

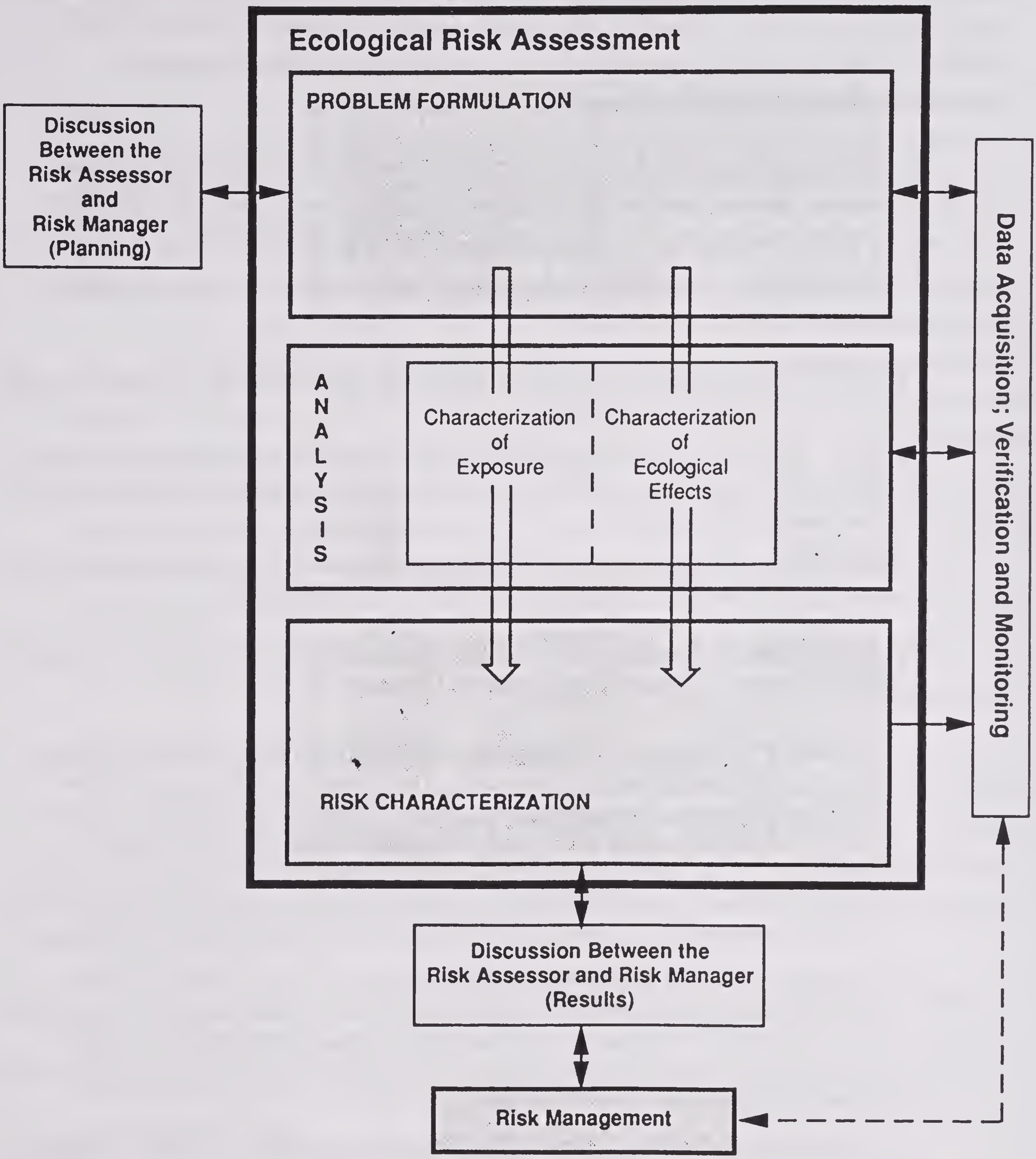


Figure 4-2. Conceptual Framework for Ecological Risk Assessment

Source: EPA, 1992a.

Models for use in ecological risk assessment are currently in developmental stages, although for some ecological components models do not yet exist. For an overview and detailed descriptions of specific models and approaches that may be applicable for use in ecological risk assessments referr to EPA (1992a and 1992b).

Strengths

Among the strengths of site-specific ecological risk assessment is that it provides the most ecologically relevant understanding on the existence of or lack of chemical-based impacts to ecosystems (SETAC, 1993). Models and methods used in ecological risk assessment have been developed and refined for a number of years in a variety of different fields, and analysts can draw on the considerable amount of practical experience in the use of these methods.

Weaknesses

In addition to the large resource requirements needed to perform a comprehensive ecological risk assessment, the results of a comprehensive study may not lend themselves to an analysis of alternative production systems. In addition, virtually all of the existing studies relate to specific sites with widespread environmental contamination from past disposal practices or the potential for future environmental contamination (SETAC, 1993).

Relevance to Impact Assessment

Because of both the technical and resource requirements needed to perform a comprehensive ecological risk assessment, its use in impact assessment would most likely be limited to LCAs of a reduced scope or be used to assess critical impact areas identified in a less detailed analysis after being triggered by the outcome of generic exposure/effects modeling efforts.

A study to evaluate the applicability of the methods and models used in ecological risk assessment (see EPA, 1992a and 1992b) would allow for a better understanding of the possible linkages between ecological risk assessment and LCA, but this issue may not be considered high priority for the following reasons:

- ecological risk assessment methods are already being refined for other purposes,
- the use of this level of detail (i.e., Tier 5) would be rare in an impact assessment, and
- those people performing risk assessments are already familiar with the basic methods.

4.12 HUMAN HEALTH RISK ASSESSMENT

Site-specific exposure/effects assessment can be accomplished through using traditional risk assessment methodology, which includes the following four components:

1. **Hazard Identification**: involves gathering and evaluating toxicity data on the types of health injury or disease that may be produced by a chemical and on the conditions of exposure under which injury or disease occurs. It may also involve characterization of the behavior of a chemical within the body and the interactions it undergoes with organs, cells, or even parts of cells. Data of the latter type can be valuable in answering the ultimate question of whether the forms of toxicity known to be produced by a chemical agent in one population group or in experimental settings are also likely to be produced in the human population group of interest. Note that risk is not assessed at this stage; hazard identification is conducted to determine whether and to what degree it is scientifically correct to infer that toxic effects observed in one setting will occur in other settings (e.g., are chemicals found to be carcinogenic or teratogenic in experimental animals also likely to be so in adequately exposed humans?).

2. **Dose-Response Assessment:** involves describing the quantitative relationship between the amount of exposure to a chemical and the extent of toxic injury or disease. Data are derived from animal studies or, less frequently, from studies in exposed human populations. A chemical agent may have many different dose-response relationships depending on the conditions of exposure (e.g., single versus repeated exposures) and the response (e.g., cancer or birth defects) being considered.

3. **Exposure Assessment:** involves describing the nature and size of the various populations exposed to a chemical agent and the magnitude and duration of their exposures. The evaluation could concern past exposures, current exposures, or exposures anticipated in the future.

4. **Risk Characterization:** involves integrating the data and analyses involved in the other three steps of risk assessment to determine the likelihood that the human population of interest will experience any of the various forms of toxicity associated with a chemical under its known or anticipated conditions of exposure (Environ, 1988).

The final step in human health risk assessment, risk characterization, is designed to generate several types of risk estimates from the results of the first three steps. Since a risk assessment typically focuses on one or two adverse human health effects, it does not reflect the full range of adverse effects of the agent or agents in question. Various choices for risk measures exist, as shown in Table 4-14. The risk measure chosen is based on how the risk assessor collects and organizes information as well as the needs of decisionmakers.

TABLE 4-14. MEASURES OF RISK FOR HUMAN HEALTH RISK ASSESSMENT

Risk Measure	Calculation	Description
Individual Lifetime Risk	dose • potency	The excess (or increase in) probability that an individual will experience a specific adverse effect as a result of exposure to a risk agent.
Population Risk	(individual lifetime risk) • (population exposed)	The number of cases resulting from one year of exposure, or the number of cases occurring in one year's time.
Relative Risk	(incidence rate in exposed group) ÷ (incidence rate in non-exposed group)	The risk in the exposed population compared to the unexposed (or differently exposed) population.
Standardized Mortality or Morbidity Ratio	(incidence rate in exposed group) ÷ (incidence rate in general population)	The number of deaths or cases of disease observed in an exposed group divided by the number expected.
Loss of Life Expectancy	(individual lifetime risk) • 36 years where 36 years = average remaining lifetime	The days or years of life lost due to a particular exposure or activity.

Source: CEQ, 1989.

The risk characterization step of human health risk assessment contains a number of areas where decisions need to be made. Some key decision areas might include the following:

- What are the statistical uncertainties in estimating the extent of health effects? How are these uncertainties to be computed and presented?
- What are the biological uncertainties in estimating the extent of health effects? What is their origin? How will they be estimated? What effect do they have on quantitative estimates? How will the uncertainties be described to decisionmakers?
- Which dose-response assessments and exposure assessments should be used?
- Which population groups should be the primary targets for protection and which provide the most meaningful expression of the health risk? (National Research Council, 1983)

Strengths

Among the strengths of site-specific human health risk assessment is that it provides the most relevant understanding of the existence of, or lack of, chemical-based impacts to human health. Models and methods used in human health risk assessment have been developed and refined for a number of years in a variety of different fields, so analysts have a considerable amount of practical experience in using these methods.

Weaknesses

The assumptions regarding the shape of the dose-response curve (e.g., linear versus nonlinear) and the existence of thresholds below which no impact occurs can have a dramatic effect on the final impact level. For example, no impact will be estimated if the ambient concentration associated with a particular emission source is under the threshold (i.e., the highest value at which no adverse health impacts can be associated with a pollutant). Dose-response functions are not always available for some impacts of concern. For instance, human health impacts associated with regulated air pollutants usually have fairly well-documented dose-response curves, but other impacts of air pollution (e.g., damage to exposed building materials) have not been fully investigated.

In addition, the analyst must estimate the population and/or resources at risk to exposure. This may be as simple as estimating the number of people living in the locale being used in this case study. However, the exercise can become more complex if only certain portions of the human population are affected (e.g., asthmatics, children). For ecosystem and natural resource impacts, the components at risk are often very difficult to estimate. In most cases, surveys of vegetation, aquatic populations, and exposed building materials, for example, are needed. If such information does not already exist, it must be gathered from the field, which is a very time-consuming and expensive task.

A number of different factors govern the degree of contact, or exposure, a person has with a toxic agent, including the period of time (duration) a person is exposed to the agent, the route (inhalation, dermal, ingestion, ocular, injection) of exposure, the amount of agent absorbed into the body by each route of exposure, environmental concentration of specific agents, and the tolerance of the exposed population to the agent. In addition to these factors, the risk assessor must also consider the demographic characteristics of the exposed population to determine the physiological parameters that affect exposure.

Relevance to Impact Assessment

Because of both the technical and resource requirements needed to perform a comprehensive human health risk assessment, its use in impact assessment would most likely be limited to LCAs of a reduced scope or be used to assess critical impact areas identified in a less detailed analysis after being triggered by the outcome of generic exposure/effects modeling efforts.

A study to evaluate the applicability of the methods and models used in human health risk assessment would allow for a better understanding of the possible linkages between human health risk assessment and LCA, but this issue may not be considered a high priority for the following reasons:

- human health risk assessment methods are already being refined for other purposes,
- the use of this level of detail (i.e., Tier 5) would be rare in an impact assessment, and
- those performing risk assessments are already familiar with the basic methods.

CHAPTER 5

RESOURCE DEPLETION: ISSUES AND CHARACTERIZATION METHODS

This chapter includes discussions of issues related to the depletion of natural resources and describes selected methods for characterizing resource depletion that have been discussed or presented in the context of LCA. Whereas some of the methods profiled in Chapter 4 account for the degradation of natural resources—that is, impacts to the supplies of natural resources (e.g., contamination of water supplies)—this chapter includes those methods to characterize natural resource depletion only. In the context of LCA, resource depletion has traditionally been addressed in a different manner than other types of impacts and thus has been separated from other methods of impact assessment as described in Chapter 4.

5.1 RESOURCE DEPLETION: KEY TERMS AND CONCEPTS

The term natural resources, in the context of LCA, refers to any component directly or indirectly derived from the natural environment (EPA, 1992c). Natural resources provide the basic raw materials for most production systems. They can be used as inputs to production systems in the form of raw materials or energy, or they can be altered or their value diminished as a result of outputs from a production system (EPA, 1992c). The complete life cycle of resources is illustrated in Figure 5-1.

Natural resources are typically classified as either stock or flow resources. This section discusses the distinction between stock and flow resources as it relates specifically to impact assessment. This distinction between stock and flow resources is important to consider because the procedures for characterizing impacts to each of these categories may be slightly different.

Stock resources include those that cannot be replenished through natural processes on time scales relevant to human societies (SETAC, 1993; EPA, 1992c). Examples of stock resources might include fossil fuels, mineral ores, surface and ground water, and soil. Stock resources are typically considered finite. In contrast, flow resources include those that can be readily replenished either by natural or artificial processes (EPA, 1992c). Examples of flow resources include most flora and fauna (e.g., trees, fish, wildlife).

Key issues associated with considering stock and flow resources in impact assessment include, but are not limited to, the following:

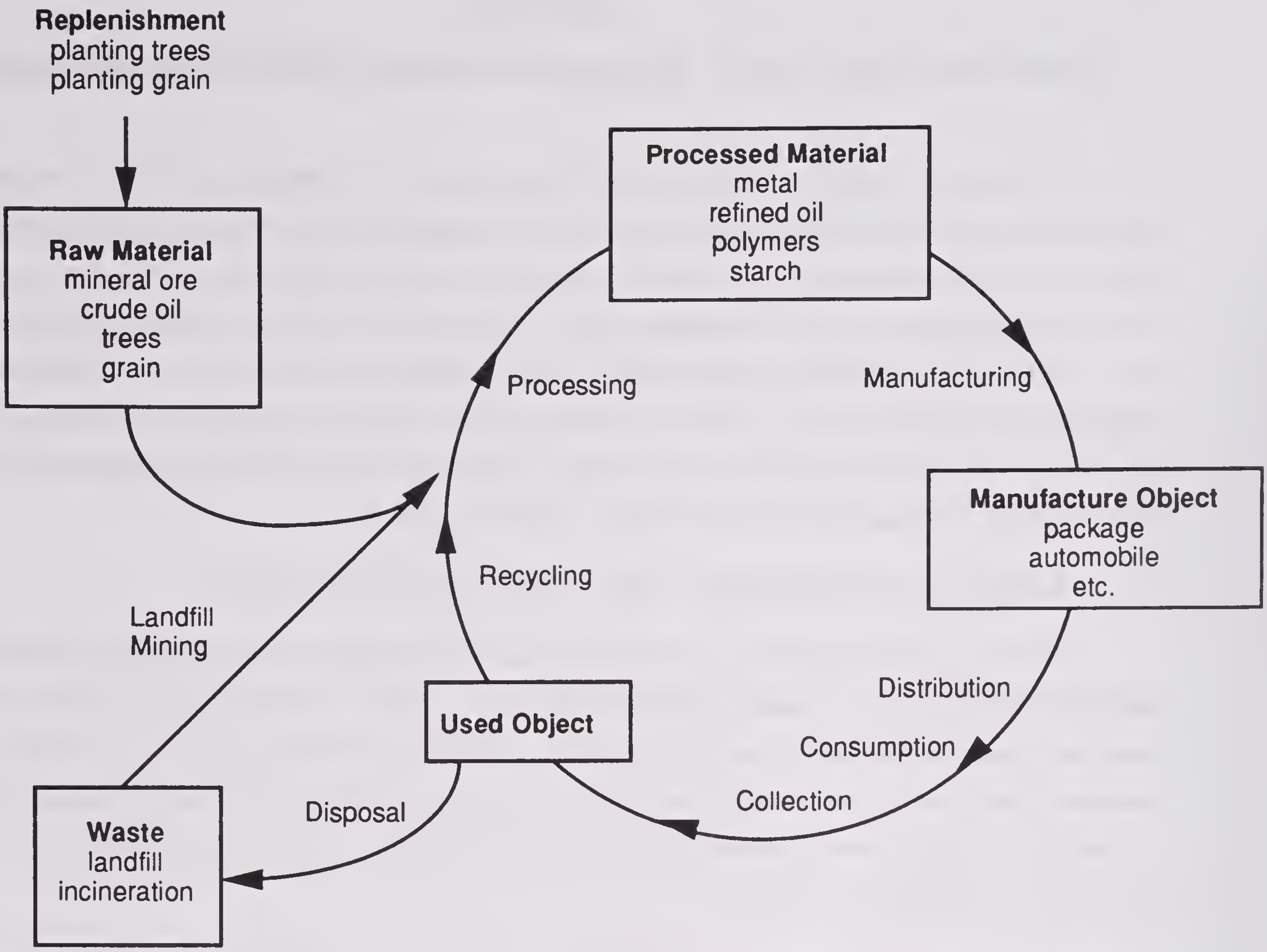


Figure 5-1. The Life Cycle of Resources

Source: EPA, 1992c

- **Base Consumption Rates:** values for the current consumption rates are governed by how clearly the resource is defined and the spatial and temporal scales within which rates are calculated.
- **Economic Factors:** levels of natural resource reserves are governed by the supply of (e.g., higher resources prices may allow for increased exploration of reserves) and demand for (e.g., lower resources prices typically lead to higher rates of consumption) natural resources.
- **Substitutability:** the use of substitute materials can preclude or reduce the rate of depletion of natural resource reserves.

- **Induced Consumption:** because the state of depletion of various resources can change over time, the magnitude and timing of induced consumption should be considered both before and after the recommendations included in the LCA are implemented.
- **Intrinsic Renewability Rates:** the growth rates for various flow resources change over time (due to, for example, increased fertilization) and must be characterized and compared to their maximum growth rates limited by the organism (SETAC, 1993).

5.2 SUSTAINABLE DEVELOPMENT AND ITS RELATIONSHIP TO RESOURCE DEPLETION

The concept of sustainable development is central to any evaluation of the depletion of natural resources—both stock and flow. The term "sustainable development" came into widespread use in 1987 when the World Commission on Environment and Development (1987) released its report *Our Common Future*, in which "sustainable development was defined as "development that meets the needs of the present generation without compromising the ability of future generations to meet their own needs."

Since then, sustainable development has taken on a multifaceted definition embodied in a process of development that achieves the following goals: 1) a level of per-capita consumption sustainable for an indefinite period of time; 2) distributional equity; 3) environmental protection, including protection of biological diversity and the continued functioning of complex natural systems; and 4) participation of all sectors of society in decisionmaking (Ascher and Healy, 1990).

Although the concept of sustainable development is relatively simple to understand, translating the seemingly simple concept into practice is still confusing. According to Ruckelshaus (1989), achieving a state of sustainable development would embody the following beliefs:

1. *The human species is part of nature.*

   Its existence depends on its ability to draw sustenance from a finite natural world; its continuance depends on its ability to abstain from destroying the natural systems that regenerate this world. This seems to be the major lesson of the current environmental situations as well as being a direct corollary of the second law of thermodynamics.

2. *Economic activity must account for the environmental costs of production.*

   Environmental regulation has made a start here, albeit a small one. The market has not even begun to be mobilized to preserve the environment; as a

consequence an increasing amount of the "wealth" we create is in a sense stolen from our descendants.

3. *The maintenance of a livable global environment depends on the sustainable development of the entire human family.*

   If 80 percent of the members of our species are poor, we cannot hope to live in a world at peace; if the poor nations attempt to improve their lot by the methods we rich have pioneered, the result will eventually be world ecological damage.

Although these beliefs seem well intended (and more or less obvious) they currently are not incorporated into organizational policymaking, unless it is in the organization's best interest to do so—such interests would generally include the realization of some benefit from changing or averting regulations or sanctions. For interests to be changed, three things are required:

- A clear set of values consistent with the consciousness of sustainability must be articulated by leaders in both the public and private sectors.
- Motivations that will support these values need to be established.
- Institutions must be developed that will effectively apply the motivations (Ruckelshaus, 1989).

From an ecological point of view, a necessary (but not sufficient) condition for sustainable development is maintaining an adequate environmental resource endowment. This endowment constitutes the natural capital (assets) necessary to provide needed and wanted environmental services—such as climate stabilization, food supply, biological waste disposal, and materials recycling.

In the context of LCA, only two long-term fates for the inputs and outputs of a production system are possible: recycling /reuse or dissipative loss. The more materials that are recycled, the less dissipation to the environment, and vice versa. Dissipative losses must be made up by replacement from virgin sources. A sustainable industrial state would therefore be characterized by minimum use of natural resources and recycling of intrinsically toxic or hazardous materials or any other materials that cause environmental problems.

5.3 RESOURCE DEPLETION MODELS

The resource depletion models described in this section are "time-metric" models. Such models are based on the basic principle that the quantity of stock or flow resource reserves (R—in units of mass) can be measured at any point in time t_i. Another class of resource depletion models is known as "value-metric" models, which generally attempt to maximize the

net value to society of any given resource consumption scheme. In essence, value-metric models can be used to estimate a benefit-cost ratio derived from producing a product versus consuming the resources required to produce the product (EPA, 1992c). Because value-metric models impress a "value" on the used resources, they may be considered valuation methods. Thus value-metric models are not discussed in this section. For a description of value-metric models, see Section 6.4 on economic valuation in this document.

The focus of this section is on time-metric resource depletion models. The key factor in time-metric models is the resource utilization rate, which is expressed as the rate of resource replenishment (dR_r/dt) minus the rate of resource consumption ($dR_{c/dt)}$ at time (t_i):

$$\text{Resource Utilization Rate} = \frac{dR}{dt_i} = dR_{rdt_i} - \frac{dR_c}{dt_i}$$

For stock resources, such as fossil fuels and minerals, the rate of resource replenishment is considered to be zero because it precludes any replenishment that is relevant to human societies. With a rate of resource replenishment equal to zero, the resource utilization rate will be negative, and any level of resource consumption will draw down, or deplete, available reserves of the stock resource. For flow resources, such as trees, the rate of resource utilization can be negative or positive, depending on whether resources are being consumed more slowly or more quickly than their rate of replenishment. When calculating the resource utilization rate, a negative value represents a net resource depletion, while a positive value represents a net resource accumulation.

Dividing the resource reserves by the rate of resource utilization yields an estimate of the time (T) until the reserves are completely depleted.

$$\text{Time Until Depletion of Reserves} = T = R/dR/dt_i$$

A positive t-value represents an accumulation of the resource, and the quantity depends on the magnitude of the t-value. A negative t-value represents resource depletion, where the magnitude of t represents the time until the resource reserves are completely exhausted.

Strengths

The majority of existing time-metric models are relatively simple to use and straightforward to understand. These models also provide a normalizing factor for aggregating resource depletion within a resource category (e.g., fossil fuels) or for comparing the depletion

of alternative resources (e.g., natural gas and oil). In addition, the time-metric models can provide an estimate of the remediability of the impact (i.e., the lower the magnitude of the t-value, the less tractable the impact).

Weaknesses

The primary weakness of the time-metric models is that they do not account for whether the replenishment of a flow resource is equal in quality to the original resource pool. For instance, although old-growth forest products represent a viable flow resource, replenishment by managed replanting will not return the forest to its original level of value or quality—at least in the near future. In addition, the time-metric models do not account for technological advances that alter the patterns of resource depletion, or for the potential substitutability of resources in the future.

Relevance to Impact Assessment

In the context of impact assessment, the depletion of a stock resource using the time-metric models would involve comparing the remaining use years with and without the product or process system, or with and without specified alternatives. In addition, any evaluation of stock-resource depletion should consider intergenerational equity or social welfare. For instance, short-term exhaustion of a stock resource would place a higher value on current populations than future populations. The analytical approach used in time-metric models allows for a clearer understanding of the distinction between stock and flow resources at local and global scales and provides the analyst with specific units for measuring resource depletion.

5.3.1 Resource Consumption Ratio

The resource consumption ratio approach characterizes the depletion of natural resources by comparing the magnitude of energy and material consumption to available supplies or reserves (EPA, 1992a). The resource consumption ratio is expressed by the following equation:

$$\text{Resource Consumption Ratio} = \frac{\text{Consumption per unit of use per unit time}}{\text{Supply per unit time}}$$

Data on the consumption of natural resources per unit use per unit time may be taken directly from the inventory analysis. Information on the supply, or reserves, of natural resources can be obtained from public or private sources (e.g., government reports, nongovernmental organizations [NGO] publications). The information obtained on the supplies of natural

resources may need to be normalized by conversion to a standard production time unit—usually annual. In addition, data on the supply of natural resources can have various measures for yields, as well as for resource reserve use rates. Different measures for yields may also need to be normalized. Table 5-1 provides examples of the application of the resource consumption ratio to various generic data.

In addition to providing a means for comparing the use of natural resources to existing supplies, the resource consumption ratio may also be used to assess the degradation of natural resources resulting from outputs or pollutants. Assessing the degradation of natural resource supplies could be accomplished by comparing the level of exposure to a pollutant to the assimilative capacity of the natural resource supply. For example, if the level of exposure of resource stock A to chemical X is 10,000 kg/year and the assimilative capacity of chemical X to resource stock A is 7,500 kg/year, then the resource consumption ratio would be 1.33. Used in this manner, a resource consumption ratio that is greater than 1 signifies that exposure to a pollutant is greater than the assimilative capacity of the resource stock and is thus a net resource degradation. A ratio that is less than 1 signifies that the resource is able to assimilate the pollutant completely. (However, this ratio does not account for exposure to multiple pollutants.)

The resource consumption ratio provides a simple means of normalizing product or process input data. The normalized figures may serve as indicators of unsustainable resource use or degradation and/or may be used to compare alternative input materials to identify those that yield minimal natural resource impacts. In addition, data on the consumption of natural resources generated in the inventory analysis may be used directly. Information on the supply, or reserves, of natural resources can be obtained from public or private sources (e.g., government reports, NGO publications).

TABLE 5-1 EXAMPLE CALCULATIONS OF GENERIC RESOURCE CONSUMPTION RATIOS

Natural Resource	Input Quantity (tons/annum)	Supply/Reserves (tons/annum)	Resource Consumption Ratio
Timber	150,000	2,600,000,000	5.7E-05
Oil	2,500	150,000,000	1.7E-05
Coal	200	500,000,000	4.0E-07
Natural Gas	575	37,500,000	1.5E-05
Iron Ore	1,350	450,000	3.0E-03

Significant efforts may be required to develop resource supply and exposure information for this approach, and it is not clear whether calculating resource consumption ratios for individual products or processes or the incremental total demand for the resource will be necessary. In addition, the significance of the resource consumption ratio is unclear.

5.3.2 Resource Depletion Matrix

The resource depletion matrix is a variation of the time-metric model that provides a conceptual framework for evaluating both the local and global depletion of stock and flow resources. This approach provides a more analytical characterization of stock- and flow-resource depletion than that obtained from inventory analyses. The more analytical approach used in this resource depletion matrix allows for a clearer understanding of the distinction between stock and flow resources at the local and global scales and provides the analyst with specific units for measuring resource depletion.

For stock resources (e.g., fossil fuels or minerals), measures of depletion are reflected by their rate of use, or exhaustion, measured in units of time. This concept is expressed by the following equation:

$$\frac{(M)}{(M/T)} = T$$

where M is mass and T is time. M represents the supply of the stock resource and theoretically has units of time. However, because the rate of production of stock resources covers such a long time span, it is assumed that the rate of production is zero.

In the depletion of flow resources (e.g., forest products or water), two attributes must be considered: (1) the size and rate of consumption of the resource "pool" and (2) the rate of replenishment (both natural and managed replacement). These two attributes are incorporated in the following equation:

$$\frac{(M)}{(M/T)} + (M/T) = (T) + \frac{(M)}{(T)}$$

where M is mass and T is time. The first term in the above equation could be used as a comparison to the depletion of stock resources because the flow resource whose current rate of consumption is greater than the rate of replenishment could be depleted in a finite period of time

if not redirected by management intervention. For example, over-harvesting of certain species of trees (e.g., mahogany in tropical forests), where the rate of consumption exceeds the rate of replenishment, will result in the depletion of the resource in a measurable period of time.

The framework for a resource depletion matrix is illustrated in Figure 5-2. This matrix is divided into four quadrants based on four categories of resources: stock, flow, local, and global. The cells corresponding to stock resources yield a measure of the time until the resource stock is depleted. The cells corresponding to flow resources provide a measure of resource depletion, which may be used to determine the sustainability of the resource use.

Characterizing flow resources is somewhat more complicated because the rate of replenishment must be considered. In addition, it is not clear whether the replenishment of a flow resource is equal in quality to the original resource pool. For instance, although old-growth forest products represent a viable flow resource, replenishment by managed replanting will not return the forest to its original level of value or quality—at least in the near future.

In the context of impact assessment, using the resource depletion matrix would involve comparing the remaining use years with and without the product or process system, or with and without specified alternatives. In addition, any evaluation of stock-resource depletion should consider intergenerational equity or social welfare. For instance, short-term exhaustion of a stock resource would place a higher value on current populations than future populations. The more analytical approach used in this resource depletion matrix allows for a clearer understanding of the distinction between stock and flow resources at the local and global scales and provides the analyst with specific units for measurement of resource depletion.

	Stock	Flow
Local	$S_L / C_L = U_L$	$C_L / R_L = D_L$ if $D_L < 1$, then $C_L - R_L = E_L$ and $Q_L / E_L = U_L$
Global	$S_G / C_G = U_G$	$C_G / R_G = D_G$ if $D_G < 1$, then $C_G - R_G = E_G$ and $Q_G / E_G = U_G$
C = consumption rate (amount/unit of time) S = stock (amount) U = use-years (time) D = depletion index (dimensionless) R = replenishment rate (amount/unit of time) E = excess consumption rate (amount/unit of time) Q = standing quantity (amount)		

Figure 5-2. Resource Depletion Matrix

Source: SETAC, 1993

CHAPTER 6

METHODS FOR CONDUCTING VALUATION

The valuation phase of impact assessment involves assigning relative values or weights to impacts based on their associated descriptors as derived in the characterization phase and stakeholder values. The primary objective of this valuation exercise is to integrate information on environmental impacts with stakeholder values to establish the relative importance of impacts or categories of impacts. Thus the challenge to practitioners is to adequately capture and express to decisionmakers the full range of potential impacts relevant to the LCA and to the stakeholders without overwhelming their audience with information.

Making successful decisions based on impact assessment requires considering all assessment results and technical information. In addition, decisions are not solely based on the precision of measurement but also on how measurements are interpreted in terms of imprecisely understood goals and values. Although developing a truly objective method for valuation is both impossible and inappropriate, several conceptual and methodological approaches to valuation do exist. Those approaches that have been used, presented, or discussed in the context of LCA are described in this chapter. In addition to the approaches described in this chapter, several integrated approaches, as discussed in Chapter 7, also contain implicit or explicit valuation components.

6.1 DECISION ANALYSIS USING MULTI-ATTRIBUTE UTILITY THEORY (MAUT)

Simply stated, decision analysis is a method that breaks down complex decisions involving multiple issues into constituent parts or individual attributes to provide a better understanding of the main factors guiding the decision. Decision analysis using MAUT is useful when deciding between largely different types of considerations. In addition, it provides a logical structure for analyzing complex weighting issues.

The first step in decision analysis is to identify all important objectives and attributes. While this step may seem obvious, it is necessary to ensure that the valuation focuses on the right problem. The objectives and attributes of the decision at hand may be identified by using tools such as an objectives hierarchy (Keeney and Raiffa, 1976). Developing an objectives hierarchy may proceed in either a top-down or bottom-up fashion:

Top-down: The decisionmaker(s) is asked to identify overall objectives. For LCA these might be to minimize overall environmental and human health impacts or to maximize public opinion.

Bottom-up: An exhaustive list of specific attributes of concern to the decisionmaker is initially identified. Items in the initial list of attributes may then be aggregated, eliminated, or redefined in the determination of a final set of attributes. For example, an initial LCA attribute list might include acid deposition impacts, solid waste impacts, corporate image, waste disposal cost minimization, etc. The decisionmaker(s) may decide that acid deposition is not a significant problem in the region and thus eliminate it from the list, resulting in a streamlined set of attributes.

Whether the objectives and attributes are determined through a top-down or bottom-up approach, the final set of attributes should have certain characteristics. An overall objective would be at the top and a comprehensive set of issue-specific objectives are then derived that are consistent with the overall objective. Finally, attributes that are meaningful, measurable, and predictable are derived for each specific objective. According to Keeney and Raiffa (1976), who describe the entire MAUT process in detail, the set of attributes should be

- comprehensive,
- as small as possible in number,
- nonoverlapping,
- judgmentally independent, and
- operational.

Decision analysis with multiple issues or objectives, such as impact assessment, would include the following steps:

1. Break the issue or decision down into single objectives and attributes.
2. Utilize the attributes to measure the degree to which an objective is achieved by a management option (attributes should be relevant to the issue, measurable, predictable, comprehensive, and nonoverlapping).
3. Identify objectives and attributes that build consensus about the nature of the issue at hand.
4. Estimate the effects of various actions (decisions) on the attributes.

An example decision tree outlining objectives and measurable attributes of water pollution effects as part of the overall objective of environmental improvement is shown in Figure 6-1. The attributes (e.g., predicted effect on human health) as shown in Figure 6-1 provide a foundation upon which analysts can estimate the effects of various actions.

In the context of impact assessment, where tradeoffs between impacts to ecosystems, human health, and natural resources must be made, employing decision analysis does not necessarily require following the above-outlined steps. Decision analysis in impact assessment would likely include employing a model to predict ecosystem, human health, and natural resource impacts and associating each impact with a unit of measure or value.

Strengths

The multi-attribute analysis capabilities of MAUT allow for an evaluation of cross-sector and/or multi-media issues. For example, in using comprehensive environmental assessment

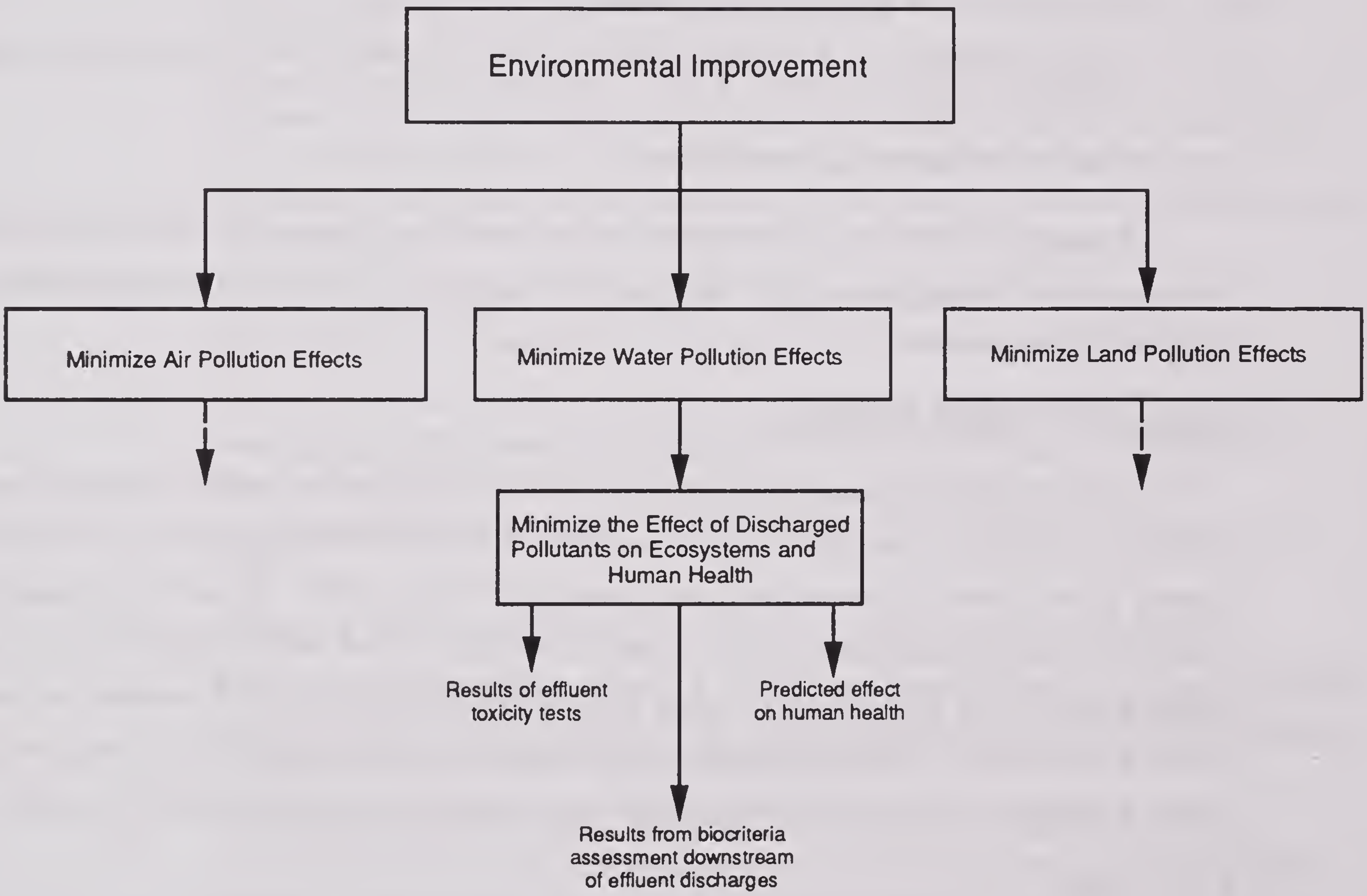


Figure 6-1. Details of MAUT Water Pollution Effects Objectives

Source: Modified from SETAC, 1993

techniques, such as LCA, decisionmakers are often faced with decisions that can cut across multiple environmental media (e.g., air pollution, water pollution, solid waste, resource use). MAUT provides a framework for breaking such multi-attribute decisions into a set of measurable attributes from which the analyst can develop a multi-attribute utility function. This multi-attribute utility function can, under favorable conditions (see Keeney and Raiffa, 1976), be broken down into single attribute utility functions, which can then be combined in a multiplicative or additive manner according to the values of estimated scaling coefficients (SETAC, 1993).

Weaknesses

The primary weakness of MAUT is that it is very difficult to implement because of some of the following characteristics:

- determination of the appropriate utility function to employ,
- decomposition of the multi-attribute utility function,
- derivation and use of multiplicative functions to combine the single attribute utility functions, and
- estimation of scaling coefficients.

Because of limiting characteristics such as those listed above, the MAUT process has been simplified and refined in "spin-off" methods such as the Analytic Hierarchy Process (AHP) described in Section 6.2.

Relevance to Impact Assessment

In the context of impact assessment, MAUT could be used to scale predicted impacts on a 0 to 100 utility scale, multiplied by the importance weights, summed, and then compared to identify the maximum utility management strategy (SETAC, 1993). However, the subjective nature of the scaling process is open to considerable debate, and analytic difficulties of the scaling process may limit whether scaling may be accomplished at all. Therefore, the most practical application of MAUT for purposes of impact assessment may be for decisionmakers to consider, separately, the importance weights and the impacts evaluation (SETAC, 1993).

6.2 AHP

The AHP is a systematic procedure for demonstrating a problem in a hierarchical structure, based on the values of the decisionmaker(s). The AHP organizes basic reasoning by

decomposing a problem into its constituent parts and then using simple pairwise comparisons to develop priority rankings in each hierarchy.

Steps to follow when using the AHP are described below. Particular steps may be emphasized more in some situations than in others, and as noted, interaction is generally necessary.

1. Define the problem and determine what you want to know.
2. Structure the hierarchy from the top (the objectives from a general viewpoint) through the intermediate levels (criteria on which subsequent levels depend) to the lowest level (which usually is a list of the alternatives).
3. Construct a set of pairwise comparison matrices for each of the lower levels—one matrix for each element in the level immediately above. An element in the higher level is said to be a governing element for those in the lower level. In a complete simple hierarchy, every element in the lower level affects every element in the upper level. The elements in the lower level are then compared to one another based on their effects on the governing elements above. This yields a square matrix of judgments. The pairwise comparisons are made based on which element dominates another. These judgments are then expressed as integers. If element A dominates element B, then the whole number integer (or exact value with decimals if known) is entered in row A, column B, and the reciprocal (fraction) is entered in row B, column A. If element B dominates element A, the reverse occurs.
4. N(n-)/2 judgments are required to develop the set of matrices in Step 3. (Reciprocals are automatically assigned in each pairwise comparison.)
5. Having made all pairwise comparisons and having entered the data, the consistency is determined using the eigen value. ($Aw = l_{max}$ w is determined. The consistency index uses the departure of l_{max} from n compared with corresponding average values for random entries to yield the consistency ratio CR).
6. Steps 3, 4, and 5 are performed for all levels and clusters in the hierarchy.
7. Hierarchical composition is used to weight the eigen vectors by the weights of the criteria and the sum is taken over all weighted eigen vector entries corresponding to those in the next lower level of the hierarchy.
8. The consistency of the entire hierarchy is determined by multiplying each consistency index by the priority of the corresponding criteria and adding them together. The result is then divided by the same type of expression using the random entry corresponding to the dimensions of each matrix weighted by the priorities as before, so that the CR is about 10 percent or less. If the CR is not 10 percent or less, the quality of the judgments should be improved, perhaps by

revising the manner in which questions are asked in making pairwise comparisons. If this fails to improve consistency, it is likely that the problem should be more accurately structured by grouping similar elements under more meaningful criteria. A return to Step 2 would be required, although only the problematic parts of the hierarchy may need revision.

9. To perform absolute measurement that preserves the rank of the alternatives and satisfies expectations and prior commitments, each lowest level subcriterion is divided into a complete set of intensities so that an alternative always reflects one of these intensities. Then the intensities are pairwise compared according to perceived importance or priority with respect to that criterion. Finally, the alternatives are rated one at a time. The intensities for each criterion and the weighted ratings are added to obtain an overall rank on a ratio scale. Unlike paired comparisons, the process to rate intensities requires expert knowledge. In most decision problems about the future, there is no such expert knowledge. Also, experts have been known to have biased and misjudged the importance of the intensities. In such cases paired comparisons must be used (Saaty, 1992).

Applying the AHP approach to the valuation phase of impact assessment is relatively straightforward. In the AHP example illustrated in Figure 6-2, the overall goal of the LCA (environmental improvement) is at the top of the hierarchy; factors affecting this goal are on the next level. These factors would probably be the impact descriptors formed in the characterization phase. Subcriteria at the next level might include economic considerations, uncertainty, assumptions, judgments, etc.

Strengths

The main strength of the AHP is that it provides an efficient framework and procedure for making individual or group decisions on single or multiple attribute problems. Some additional strengths of the AHP include the following:

- relatively simple and straightforward to use,
- available AHP computer software package (called Expert Choice),
- overall view of complex relationships inherent in multi-faceted problems and in the judgment process, and
- flexible enough to handle a wide variety of problem types.

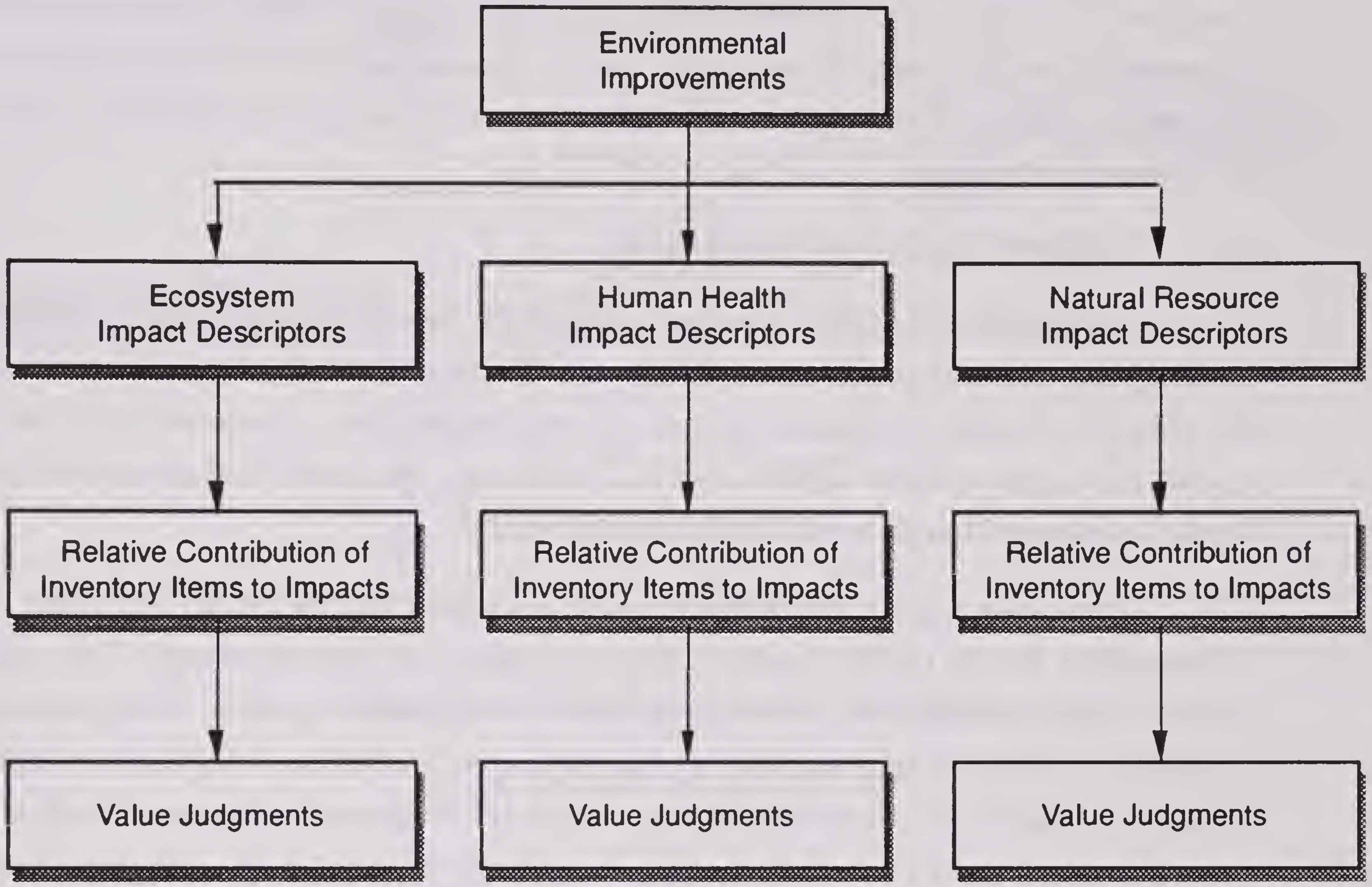


Figure 6-2. Example Framework for AHP Applied to Impact Assessment

Weaknesses

One weakness of the AHP results from the pairwise comparison process. This process requires expert knowledge to rate the intensities (see Step 9 in the AHP process outlined above) of the attributes being compared. In the case of most future problems, there is no such expert knowledge. In addition, the possibility exists that the experts can have a bias and/or might misjudge the importance of particular attribute intensities. At any rate, because of its reliance on the values and judgment of a select group of individuals, it is unlikely that the results of an AHP study could be replicated.

Relevance to Impact Assessment

The AHP may provide a useful tool for evaluating multi-attribute, complex problems. Such problems typify those encountered in the valuation phase of impact assessment where, foreseeably, a wide variety of multi-media and/or cross-sector environmental impacts must be considered. The AHP also provides a useful framework for integrating stakeholder values with environmental impacts.

6.3 MODIFIED DELPHI TECHNIQUE

The Delphi Technique is a procedure originally developed by the Rand Corporation for eliciting and processing the opinions of a group of experts knowledgeable in the various areas involved. The Delphi Technique addresses the need to structure a group communication process to obtain a useful result for a given objective. In essence, the Delphi Technique attempts to create a structured format to elicit collective knowledge.

In response to a number of shortcomings associated with the Delphi Technique (see Linstone and Turoff, 1975), a modified Delphi technique has been developed. This modified Delphi technique provides a systematic and controlled process of queuing and aggregating the judgments of group members and stresses iteration with feedback to arrive at a convergent consensus. The weighting system discussed in the following section does not include all the elements of the original Delphi Technique. In addition, results of these ranking sessions need further study, feedback, and substantive input from field data before using.

The weighting procedure can be simply employed. A deck of cards is given to each person participating in the weighting. In this example each card names a different technical specialty. Each of the participants is then asked to rank the technical specialties according to their relative importance to explaining changes in the environment that would result from a particular system. Then each individual is asked to review the list and make pairwise comparisons between technical specialties, beginning with the most important specialty. The most important technical specialty is compared with the next important specialty by each individual, and the second technical specialty with respect to the first. For example, the first technical area might receive a weight of 100 percent, and the second most important technical area might be considered only 90 percent as important as the first. The second and third most important technical specialties are compared, and the third most important is assigned a number of—for example, 95 percent—based on its relative importance compared to the second most important technical specialty. A sample diagram of the comparison is presented in Figure 6-3.

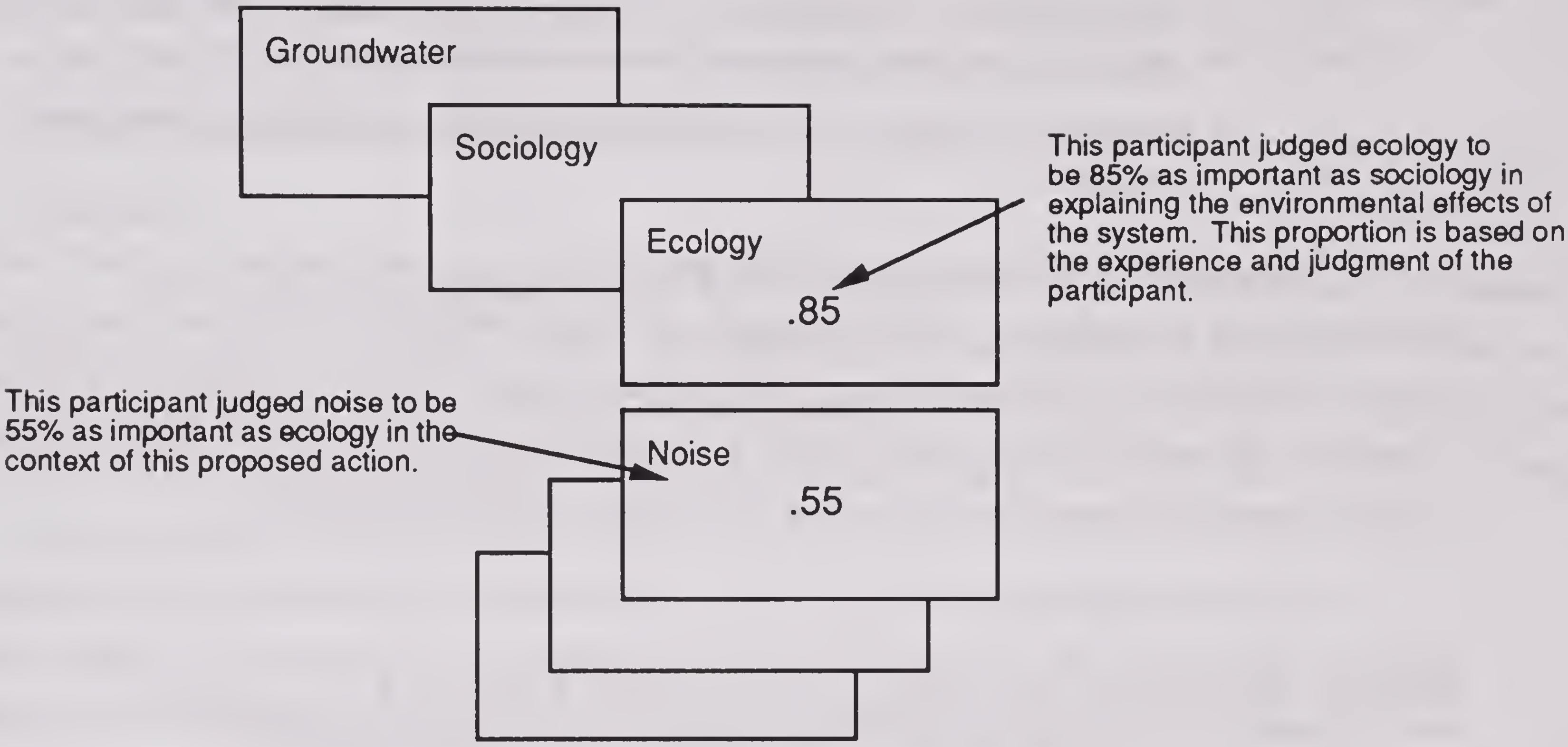


Figure 6-3. Modified Delphi Technique

Source: Modified from Jain et al., 1993.

The formula for weighting the technical specialties is

$$W_{ij} = \frac{V_{ij}}{\sum_{i-1}^{n} V_{ij}} P \qquad (i = 1,2,3,\ldots,n)$$

$$V_{ij} = \begin{cases} 1 & (i = 1) \\ V_{i-1j} X_{ij} & (i = 2,3,\ldots,n) \end{cases}$$

where

W_{ij} = weight for the ith technical specialty area by the jth scientist,

n = number of technical specialties,

P = 1,000: total number of points to be distributed among the technical specialties,

X_{ij} = the jth scientist's assessment of the ratio of importance of the ith technical specialty in relation to the (i-1)th technical specialties, and

V_{ij} = measure of relative weight for the ith technical specialty area by the jth scientist.

To accomplish the second part of this technique (i.e., to rank attributes within a technical specialty), each participant or group independently ranks attributes in his or her own specialty. The information from these pairwise comparisons can then be used to calculate the relative importance of each of these specialty areas; a fixed number of points (e.g., 1,000) is distributed among the technical specialties according to individual relative importance.

After the weights are calculated from the first round of this procedure, the information about the relative weights is presented again to the experts, a discussion of the weights ensues, and a second round of pairwise comparisons is made. The process is repeated until the results become relatively stable in successive rounds.

In a demonstration of this method, an interdisciplinary group of college graduates with very little training was asked to rate the following areas according to their relative importance in environmental impact analysis and to distribute a 1,000-point total among these categories:

- air quality
- ecology
- water quality
- aesthetics
- economics
- transportation
- earth science
- sociology
- natural resources and energy
- health science
- land use
- noise

After a thorough group study of all 12 areas, the group was asked to rate the areas again. The results, shown in Table 6-1, indicate that although some relative priorities changed, the points allocated to each category remained essentially the same. Similar ratings may be developed for attributes within each group.

Strengths

This modified Delphi technique provides a systematic and controlled process of queuing and aggregating the judgments of group members and stresses iteration with feedback to arrive at a convergent consensus. Attributes within a technical specialty are ranked by an expert in that technical specialty and aggregated over the expert panel, thereby creating a structure for ranking alternative impact areas (see Table 6-1).

TABLE 6-1. EXAMPLE RESULTS OF USING THE MODIFIED DELPHI PROCEDURE FOR COMPARING ENVIRONMENTAL AREAS

Before Interdisciplinary Study		After Interdisciplinary Study	
Area	Average Point Distribution	Area	Average Point Distribution
Water	125	Water	128
Air	122	Air	126
Natural Resources	109	Natural Resources	105
Health	100	Ecology	93
Ecology	97	Health	88
Land Use	81	Earth Science	87
Earth Science	79	Land Use	78
Economics	62	Sociology	64
Sociology	60	Noise	62
Transportation	56	Economics	62
Aesthetics	54	Transportation	61
Noise	53	Aesthetics	46
TOTAL	1,000	TOTAL	1,000

NOTE: The numeric values in this table are particular to a specific case study. A different group would certainly arrive at different decisions, and any application directed toward comparison between attributes should be made in the context of a specific planning situation.

Source: Jain et al., 1993.

Weaknesses

One of the weaknesses of the modified Delphi technique is one that typically plagues most valuation tools—namely the requirement of expert knowledge with which to rate environmental attributes. In many cases, there is no such expert knowledge. In addition, the possibility exists that the experts can have a bias and/or may misjudge the importance of particular attribute intensities.

Another weakness of the modified Delphi technique is in the ranking process. This process requires a wide variety of technical specialists to rank attributes within their respective technical specialty area. In addition, the results of the ranking sessions may require further study, feedback, and substantive input from field data before using. Conceivably, a large amount of time and resources could be spent on such follow-up analysis.

Relevance to Impact Assessment

The information generated from the modified Delphi technique may provide a useful procedure for calculating the relative importance of each specialty (i.e., environmental attributes or impacts) area. From this, a fixed number of points (e.g., 1,000) may be distributed among the technical specialties, thus indicating the relative importance of individual specialty areas. However, the level of technical expertise and time required to conduct a thorough evaluation of each specialty area may limit the application of the modified Delphi technique to valuation.

6.4 LIFE-CYCLE COSTING[1]

A life-cycle inventory would address environmental inputs and outputs of a production system, while the impact assessment would address the environmental impacts associated with those inputs and outputs. Life-cycle costing extends impact assessment by taking an additional step (i.e., placing a dollar value on impacts). Methods for assigning costs are described below.

Monetary values for environmental impacts can be determined for certain types of impacts. The market value, for instance, of crop loss or damages caused by air pollutants can be valued directly by assessing the market value of the lost output. However, quantifying an impact chain leading to revenue loss may be difficult. For example, translating NO_x emissions from the production of a glass bottle into an incremental change in ambient ozone concentration, and quantifying crop loss from that increment is highly uncertain. In addition, placing monetary

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[1]Portions of this section were summarized from White et al. (forthcoming).

values on many impacts (e.g., adverse human health effects) is difficult from an economic and ethical perspective.

For the purpose of analysis, different types of value that individuals place on the environment have been distinguished: use value, option value, and existence value. Use value is based on the utility people derive from the "consumption" of the environment for recreational purposes, such as boating, fishing, and other sporting activities. The option value is the use value in the presence of uncertainty. People may not consume the environment at present but may want to do so in the future. Having the option for future use is assumed to be valued by consumers. Finally, the existence value is the value people assign to the environment for "altruistic" reasons; it is the utility they derive from the knowledge of the existence of the environment.

Several methods are available for indirectly valuing impacts by estimating the use, option, and/or existence values that individuals place on environmental amenities or the devaluation resulting from environmental harm. These methods involve the following:
1) examining behavioral responses that are, or might be, influenced by an externality;
2) assuming or creating a fictitious market to elicit the value that individuals might assign to an externality; or 3) analyzing the implicit value placed on pollution abatement by society through the actions of its regulatory agencies. Methods in each of these three categories are briefly described below. For detailed descriptions of these methods and their corresponding strengths and weaknesses, the reader is referred to White et. al. (Forthcoming), Tellus Institute (1992a, 1992b, 1992c), Desvouges et al. (1991 and 1989), and the Organization for Economic Co-operation and Development (OECD) (1989).

Strengths

One of the main strengths of life-cycle costing is that the basis for measurement (i.e., dollars) is a metric that most people can readily understand. Monetary values for environmental attributes also enable analysts and decisionmakers to directly compare environmental and economic considerations, whereas environmental and economic decisionmaking have generally been treated as separate, unrelated entities.

Another strength of life-cycle costing is that the valuation methods and techniques have been refined over a long period of time, are applicable to a wide variety of impact types, and offer much practical experience for analysts to draw upon.

Weaknesses

Life-cycle costing is open to criticism for using economic valuation methods to "price" environmental attributes (e.g., the extinction of species, loss of pristine forest habitat, or adverse human health effects). For example, a comprehensive estimate of society's willingness to accept the loss of the spotted owl in the Western United States can easily surpass the GNP of most, if not all, countries. Some additional criticisms of using monetary values to assess environmental impacts are the following:

- large gap between rich and poor in terms of disposable resources for environmental care,
- needs of today often outweigh the needs for tomorrow,
- insufficient knowledge to value environmental impacts because the full consequences of impacts are not fully understood, and
- monetary valuation focuses on human needs.

In addition, methods of life-cycle costing often rely on a set of assumptions that may or may not accurately reflect reality. Some of these assumptions are outlined in the discussion of specific methods below.

Relevance to Impact Assessment

Life-cycle costing methods may be useful in the context of impact assessment for translating impacts into a common metric (i.e., dollars) for direct comparison of impacts within and between impact categories. The presentation of impacts in monetary terms also can facilitate decisionmakers' consideration of tradeoffs between environmental and economic issues.

One integrated approach to impact assessment—the EPS Enviro-Accounting Method outlined in Chapter 7—provides an example of the use of economic valuation in the context of LCA.

6.4.1 Hedonic Pricing

Hedonic pricing attempts to identify a surrogate for the nonexistent market for the environment. Markets that qualify as surrogate markets for the environment are those in which a private good is traded that may bear some relationship to the public environmental good. The notion underlying the concept of hedonic prices is that people derive utility from various

attributes of a product. A product has many attributes, some of which can relate to the presence of a public good. A house, for example, can have features individual consumers value differently. Each of these common features commands a price; however, this price is implicit: individual features of a house are not sold separately. One attribute of the house is the environment in which it is located.

In theory, one can construct demand functions that depend on these individual characteristics, and one can derive the amount of money consumers are willing to spend to obtain one more unit of q, the environmental quality feature. (If q is air quality, then "one more unit of q" would refer to "one unit less of pollutant," where the "pollutant" could refer to an index of air pollution.) One would expect to observe differentials in housing prices, depending on the quality of the specific environment in which they are located.

The notion of a good embodying many characteristics implies that a job, too, has many characteristics in addition to the wage that it pays. One important characteristic is the risk to the health and life of the worker. It is argued that workers will only accept a job with high risk when given a "compensating wage differential." The hedonic wage method relates the size of wage differentials for various jobs to their lives.

For this approach to measure what it intends to measure, several assumptions must be made pertaining to the aggregability of individual preferences (see OECD, 1989). In addition, it is subject to many sources of bias (see OECD, 1989), for example, strategic bias. Because environmental quality is a public good (once it is provided, people cannot be excluded from its consumption), people have an incentive to understate their preference (if they are held to pay), counting on the fact that other people will provide for the supply of the good. This is the free-rider problem. Also several sources of bias are based on individual rationality. It has been observed that people respond to the starting value that is quoted to them (source for the "starting point bias"). In addition, there is also concern about whether the hypothetical markets correspond well enough to real markets.

Apart from various technical problems (see OECD, 1989), the obvious flaw of this approach is that it only targets the value of an area for a very specific narrow use. Surely people value natural resources for more than the amenities they offer. And again, there is no way this method would allow the contribution of a single pollutant to environmental degradation to be evaluated.

The derivation of an implicit price for an environmental characteristic from an ideal type demand function is rarely a straightforward calculation. Estimating these implicit processes from observable market data, however, requires strong assumptions and is not without problems. Apart from the usual assumptions about the structure of individual utility functions relating to aggregability, it has to be assumed that people have a wide enough array of choices to make their decisions on the basis of all characteristics. This is obviously hardly ever the case. Often, one characteristic overrides all others; proximity to the place of work often takes this role. People do not usually have a choice about where they find work; thus, they may move into an environment that they would not move into otherwise.

Another problem is that finding a sample with sufficient variation (i.e., enough houses that exhibit different characteristics) is not easy. The specific environment of houses varies together with other factors, and it is very hard to isolate the influence of one variable when they vary together. And, as stated above, in the absence of a wide array of choices, people are likely to base their decisions on characteristics other than the environment.

One problem with this method is that it presupposes information about job characteristics on the part of workers and researchers. Workers often do not have sufficient information about the risks to their health and life posed by their jobs. Also, unless a job exposes one to specific pollutants, establishing a worker's dislike for a specific pollutant is not possible. This method also involves the problem of measurement. Data on specific pollution at work are not readily available; data usually only exist on the consequences of hazards, such as accidents, morbidity, and mortality. Hedonic wage studies would be more useful in damage cost studies if they could indicate the value that people ascribe to their lives.

6.4.2 Contingent Valuation

Contingent valuation assumes hypothetical (contingent) markets. In essence, it consists of experiments in which people are asked to express their valuation for a specific environmental commodity. These experiments can be designed as bidding games, questionnaires, and so forth (see Freeman, 1982 and Mitchell and Carson, 1991).

Understanding the change in environmental conditions consumers are asked to evaluate is important. Two concepts are suggested in the literature: willingness to pay (WTP) and willingness to accept (WTA). Loosely speaking, the former is the amount of money a consumer would be willing to spend to secure an environmental benefit, and the latter is the compensation that the consumer would demand to accept an environmental cost. However, both concepts can

be applied to similar changes in environmental conditions. For example, consider a policy to clean up 90 percent of sulfur oxides emissions. WTP then is the maximum amount of money an individual would give away to have 90 percent of the sulfur oxides emissions abated, while maintaining his or her utility level, and WTA is the amount of money he or she would have to be given to accept the pollution while maintaining the utility level corresponding to the absence of 90 percent of the present pollution.

Economic theory suggests that these two values do not really differ. However, empirical studies assessing the magnitude of WTA versus WTP have consistently produced far greater amounts for WTA than for WTP. There has been ongoing discussion about this apparent discrepancy. It was long known that the greater the difference between WTA and WTP, the greater the income elasticity of demand. WTP is obviously limited by an income constraint, whereas WTA is not.

6.4.3 Cost of Control Valuation

The cost of control valuation method enjoys increasing popularity as utility companies attempt to internalize the environmental cost of energy production. Some states (e.g., California, Massachusetts, Nevada, New York, Wisconsin, Oregon) have proposed or adopted this approach to incorporate the environmental costs of electricity production into their energy planning processes.

This approach infers that the cost society attributes to pollution may be derived from government regulations for specific pollutants. Complying with standards set for pollutant emission is costly; thus, there must be a perceived benefit to pollution abatement. Two concepts are central to this approach: the marginal cost of pollution abatement and the marginal benefit of pollution abatement.

- **Marginal Cost of Pollution Abatement** is an increasing function of the amount of pollutant being controlled. Increasing marginal cost also means that the unit cost of abatement (the cost of abatement per unit of pollutant) rises as more and more pollution is abated. To remove the first unit of pollutant, one would choose the cheapest technology available. The most expensive technology would only be employed if the potential of cheaper technologies was exhausted.

- **Marginal Benefit of Pollutant Abatement** is a decreasing function of the amount of pollutant being removed. For example, the benefit from preventing one more ton of SO_2 to enter the atmosphere is smaller the more SO_2 has already been controlled. The

negative side of this relationship is that the marginal damage function of pollution is generally increasing; that is, the damage that one unit of pollutant causes is greater, the higher the overall pollution levels.

The optimal emission standard for a particular pollutant is that level of pollutant at which the marginal cost of abatement equals the marginal benefit of abatement. Setting such a standard would require an efficient allocation of resources for pollution abatement activity. To do more would cost society more than the benefits that would result from implementing the standard.

Several problems are associated with the pollution abatement approaches described above. First, no emission standard exists for each individual pollutant. Controls—not standards—are administered for some pollutants; others are not regulated at all. Controls present the problem of "joint cost of pollution control": where several pollutants causing different environmental impacts can be captured with one-and-the-same device. The problem lies in how the cost of that device should be allocated to individual pollutants. In addition, a value for the pollutant the device is intended to capture can only be inferred because the regulation implies a certain value for this pollutant.

Another problem is that regulations for all pollutants may not exist. A case in point is the emission of greenhouse gases. One could value the costs of these emissions through the costs of the measures that would offset the emissions (e.g., afforestation). It also seems legitimate to assume that society holds consistent preferences, and that for some pollutants, regulations addressing different but similar ones can be used. For example, the banning of lead acid batteries from incinerators reveals the regulator's (representing society's) preference that heavy metals should not be emitted. It seems legitimate to assume a regulation banning other heavy-metal products of similar toxicity.

CHAPTER 7

INTEGRATED METHODS FOR IMPACT ASSESSMENT

Integrated methods have been developed, or are being developed, to include some combination of classification, characterization, and/or valuation activities of impact assessment. This chapter profiles some of these integrated methods. Some of these methods integrate data developed from an inventory analysis with expert decision or economic valuation methods to yield information that is relevant to not only environmental decisionmaking but also to overall business decisionmaking, which includes a number of factors (e.g., profitability, product quality) in addition to environmental performance.

7.1 IMPACT ANALYSIS MATRIX (IAM)

The IAM is an exploratory, qualitative, expert-based approach to impact analysis that builds on the results of an inventory analysis. The IAM approach is described below by explaining its development and initial use.

The IAM approach was developed as part of a broader assessment of source-reduction potential for halogenated solvents, which included an assessment of alternatives to such solvents in specific applications. The IAM allowed for the direct evaluation of the relative environmental burdens of a particular application of a halogenated solvent and its alternative(s) and made the tradeoffs between them explicit. Two specific applications involving substitution systems for TCA (1.1.1-trichloroethane) were evaluated:

- substitution of a caustic aqueous cleaner for TCA vapor cleaning of metal parts and
- substitution of supercritical CO_2 paint spraying for TCA-based paint spraying.

Comparisons of these two TCA substitute systems were conducted on two different levels: user (or shop) level and global level. User-level impacts referred to ecosystem impacts that emanated from a boundary drawn around a particular facility using the TCA substitute system. For example, only waste disposal activities associated with using the substitute were considered. Global-level impacts took into account all of the traditional life-cycle stages, including raw materials acquisition, manufacturing, use/reuse/maintenance, and recycle/waste management. Analysis at these two levels allowed for identifying additional tradeoffs between the two systems. That is, it allowed options that appeared favorable from the user's point of view but unfavorable from a global point of view, or vice versa, to be identified and evaluated.

The IAM process entails convening a group of experts to carry out the following steps:

1. Identify appropriate impact categories. The IAM study of the two TCA substitutes consisted of five columns of inventory data (inputs and outputs) and seven rows of ecosystem impact categories. These impact categories were selected by expert judgment and included

   - global warming
   - ozone depleting potential
   - nonrenewable resource utilization
   - air quality
   - water quality
   - land disposal
   - transportation effects

   (It should be noted that in applying the IAM approach in other settings, impact categories that match the particular characteristics of each comparison should be used. The identification and exclusion of various impact categories should be transparent and sufficient justification should be provided.)

2. Determine which cells in the IAM represent either double counting or meaningless comparisons. For instance, in the case of the two TCA substitutes it was determined that aqueous wastes had no significant impact on global warming; thus the corresponding cell in the IAM was eliminated. As a result of this step, 17 of the 35 IAM cells were eliminated.

3. Assign unweighted "scores" to each viable cell in the IAM. Scores for the TCA study were assigned in relation to a particular option chosen as the base. In this study, a "+" was used to signify a larger ecosystem impact than the base option (TCA), and a "–" was used to signify a lesser impact. A "0" can be used to signify little or no perceived difference in impact. Determination of scores was based on a combination of life-cycle inventory data and expert knowledge of associated ecosystem impacts.

4. (Optional). Apply weights to the initial unweighted scores to determine if the results will change significantly. The weighting scheme used in the TCA study assigned a "++" to relatively strong ecosystem impacts and a "—" to relatively large reductions in impact.

   (It should be noted that the weighting may or may not be restricted to a single impact category, depending on the views of the expert panel. However, the basis for assigning weights and the scope of comparison within and across impact categories should be made transparent.)

5. Sum the individual cell scores (pluses and minuses) to derive overall scores for each row and column and, if appropriate, for the entire matrix. Unweighted scores in the TCA substitute IAM ranged from +18 to -18, and weighted scores ranged from +36 to -36.

Table 7-1 shows data gathered for the two TCA substitutes. For each substitute, the data were broken down into user-level and global-level items. The corresponding IAMs for the TCA substitutes are shown in Figures 7-1 and 7-2. As an example of the type of information that may be derived from the IAM approach, compare and contrast the scores in the energy-inputs column evaluated at the user versus the global levels. From the perspective of the user, impacts derived

TABLE 7-1. TCA SUBSTITUTE STUDY INVENTORY DATA

	Vapor Degreasing		Aqueous Cleaning	
Parameters	**User**	**Global**	**User**	**Global**
Amount Used (tons)				
TCA	26.6	0	0	0
Aqueous cleaner	0	0	2.7	0
Material Inputs				
Trona deposits, salt, sand	0	1.2	0	4.5
Crude, natural gases	0	2.9	0	0
Energy Inputs				
Power or Fuel (per million BTU)	520	1,530	1,730	1,800
Atmospheric Emissions (tons)				
Cl-HC, HC/particulates, Cl_2	0	2	0	<0.1
TCA	21.6	21.6	0	0
Particulates	0	0	0	<0.1
Water vapor	0	0	288	288
Aqueous Wastes (tons)	0	682	1,822	1,822
Solid Wastes (including spent catalyst, solids/ sludge, used oil, and shale—in tons)	0	4.9	0	0.3
TCA and oil (from OTVD)	6.5	6.5	0	0

Source: Source Reduction Research Partnership, 1991.

User-Level Impact Analysis Matrix						
	Impacting Parameters					
Ecosystem Impact Categories	**Material Inputs**	**Energy Inputs**	**Air Emissions**	**Aqueous Wastes**	**Solid Wastes**	**TOTAL**
Global Warming		+1	-1			0
Ozone Depleting Potential			-1			-1
Stock Resource Use	-1	+1				0
Air Quality		+1	-1	+1	-1	0
Water Quality		+1		+1		+2
Land Disposal		+1		+1	-1	+1
Transportation Effects	-1	+1			-1	-1
TOTAL	-2	+6	-3	+3	-3	+1

Notes: 1. Shaded cells signify no basis for impact.
2. A rating of "-1" represents decreased impact, "0" represents the same impact, and "+1" represents an increased impact.

Figure 7-1. User-Level Impact Analysis Matrix for Ecosystem Impacts

Source: Source Reduction Research Partnership, 1991.

Global-Level Impact Analysis Matrix						
	Impacting Parameters					
Ecosystem Impact Categories	**Material Inputs**	**Energy Inputs**	**Air Emissions**	**Aqueous Wastes**	**Solid Wastes**	**TOTAL**
Global Warming		0	-1			-1
Ozone Depleting Potential			-1			-1
Nonrenewable Resource Use	-1	0				-1
Air Quality		0	-1	+1	-1	-1
Water Quality		0		+1		+1
Land Disposal		0		+1	-1	0
Transportation Effects	-1	0			-1	-2
TOTAL	-2	0	-3	+3	-3	-5

Notes: 1. Shaded cells signify no basis for impact.
2. A rating of "-1" represents decreased impact, "0" represents the same impact, and "+1" represents an increased impact.

Figure 7-2. Global-Level Impact Analysis Matrix for Ecosystem Impacts

Source: Source Reduction Research Partnership, 1991.

from energy input requirements were a dominating category and were much higher for the aqueous substitute relative to the TCA system because of the high pumping and heating requirements of the aqueous substitute. In contrast, global-level impacts derived from energy requirements were found to be essentially the same for the two systems.

Strengths

The IAM is relatively simple and convenient to use, is flexible enough to account for a wide variety of impacting parameters (i.e., life-cycle components) and environmental impact categories, and can be used at different levels of analysis (e.g., global versus shop level). The IAM also does not require any additional data beyond that which is generated in the inventory analysis and uses a relatively objective technique (i.e., less is better) to evaluate the associated environmental consequences.

Weaknesses

One weakness of the IAM is that it does not measure impacts. Appropriate impact categories are chosen by expert judgment, and inventory items from two alternatives are merely compared according to a "less is better" ranking for their contribution to their associated impact categories. However, this process does not provide insight into how impact categories relate to one another. For example, in Figure 7-2 both the totals for global warming and nonrenewable resource are -1. From this, the reduction in global warming and nonrenewable resource use appear to be "equal" from the use of aqueous cleaners. However, the method does not indicate, for example, how better or worse a 1-ton reduction in global warming gases is compared to a 1-ton reduction in nonrenewable resource use.

One possible variation to the IAM matrix that may help to better express the relationship of impact categories to one another is the Leopold interaction matrix. The cells in the Leopold interaction matrix contain the ratio of the magnitude of impact (M) to the importance of the impact (I). M expresses the extensiveness or scale of the impact, and I expresses the importance of the impact (to stakeholders). The basic framework for the Leopold interaction matrix is shown in Table 7-2.

TABLE 7-2. LEOPOLD INTERACTION MATRIX

	Life-Cycle Stage			
Impact Category	Raw Materials Acquisition	Manufac-turing	Use/Reuse/ Maintenance	Recycle/Waste Management
Global Warming	M / I	M / I	M / I	M / I
Ozone Depleting Potential	M / I	M / I	M / I	M / I
Nonrenewable Resource Utilization	M / I	M / I	M / I	M / I
Air Quality	M / I	M / I	M / I	M / I
Water Quality	M / I	M / I	M / I	M / I
Land Disposal	M / I	M / I	M / I	M / I
Transportation Effects	M / I	M / I	M / I	M / I

Scale Ranges: M → 1 to 10 1 = lowest magnitude of impact, or lowest level of importance.
I → 1 to 10 10 = highest magnitude of impact, or highest level of importance.

Another weakness of the IAM is that its use in a noncomparative study, which includes only a single set of data from one alternative and no set of data against which to evaluate the alternative, is not clear. For example, in the case study example outlined above, an IAM for aqueous cleaning alone would be meaningless without the baseline of vapor cleaning against which to compare aqueous cleaning. With just one set of data, the IAM could possibly be modified to provide a general indication of the impact categories and/or impacting parameters that are most significantly affected. In this case the pluses and minuses in the matrix cells would be used to represent the relative significance of particular impact categories or impacting parameters.

Relevance to Impact Assessment

The IAM approach may provide a relatively simple, quick, and useful means of qualitatively comparing the environmental implications of two or more alternative systems without having to characterizing impacts. The more qualitative nature of the IAM would make it more appropriate for internal applications or as a screening tool to identify impact categories or life-cycle components that require a more detailed level of assessment.

7.2 THE EPS ENVIRO-ACCOUNTING METHOD

Prepared for the Swedish Environmental Research Institute, the EPS Enviro-Accounting method describes impacts on the environment in terms of one or several safeguard subjects using the EPS method described in Chapter 4 and then places a value on changes in the safeguard subjects according to the WTP to restore them to their normal status.

The five safeguard subjects included in the EPS Enviro-Accounting method are the following:

- human health,
- biodiversity,
- production,
- resources, and
- aesthetic values (Swedish Waste Research Council, 1992).

Impacts are characterized and valued on a relative scale using ELUs according to the WTP for avoiding negative effects on the safeguard subjects. Environmental indices are then calculated for the materials and processes being studied. Background information is derived from LCA-based inventories of the materials and processes under review. The values are not absolute figures but rather points of reference for further analysis and refinement. Environmental impact valuation is described as a subjective matter that can be given some degree of objectivity by studying decisions made in society or by surveying people's opinions. Contingent valuation is cited as a method for generating a relative rating of various environmental effects. Contingent valuation is used in the EPS Enviro-Accounting approach to determine individuals' WTP to avoid certain environmental effects. To date, EPS indices for a wide range of environmental impacts have been developed using such WTP figures.

The output from the EPS Enviro-Accounting system is a value, based on a common metric, for different environmental impacts. The value may be broken down into its individual components for further analysis, and the user can determine the level of detail desired.

Strengths

The EPS Enviro-Accounting method is strong in that it accounts for a wide variety of impacts within five main impact categories: human health, biodiversity, production, resources, and aesthetic values. Impacts within and between these main categories are characterized and valued on a relative scale allowing for a direct relative comparison of impacts. In addition, the

information required in the EPS Enviro-Accounting methods is derived from LCA-based inventories and readily available environmental valuation studies.

Weaknesses

One weakness of the EPS Enviro-Accounting method is that environmental impact valuation is a highly subjective matter. Although monetary units provide an easily understood value metric, using monetary values to assess environmental impacts has been criticized for several reasons:

- a large gap exists between rich and poor in terms of disposable resources for environmental care,
- the needs of today often outweigh the needs for tomorrow,
- insufficient knowledge exists to value environmental impacts because the full consequences of impacts are not fully understood, and
- monetary valuation focuses on human needs.

Relevance to Impact Assessment

Because the EPS Enviro-Accounting method was developed in the context of LCA, it is readily applicable for impact assessment. The five safeguard subjects may be used to categorize inventory items into impacts categories, and environmental valuation studies using WTP may be used to estimate costs and develop coefficients expressing the relative environmental impact (in economic terms) of alternative items. However, environmental valuation studies are sometimes controversial in their own field of economic research. The use of such valuation studies and/or techniques for impact assessment may be similarly problematic.

7.3 INTEGRATED MANUFACTURING AND DESIGN INITIATIVE (IMDI) ENVIRONMENTALLY CONSCIOUS MANUFACTURING (ECM) LIFE-CYCLE ANALYSIS

As part of a Sandia National Laboratory program, IMDI selected Department of Energy (DOE) stakeholders (e.g., designers, manufacturers, Environmental Safety and Health personnel, environmental technology staff, industry, EPA, and academicians) and surveyed them to establish a basis for defining environmental impact metrics. A panel discussion was also conducted. The survey asked for two primary responses:

1) "Identify environmental impacts of activities related to manufacturing, use and disposal;" and

2) "… list the criteria that might be used to assess one product or process against one another with respect to minimizing those impacts" (Watkins, 1993).

The panel used the AHP process, supported by Expert Choice software, and group decisionmaking. The panel developed an IMDI Environmental Impacts Model that builds on earlier SETAC work.

The panel discussed the possibility of using Colby's (1991) five environmental management paradigms as a basis for assigning weights to environmental impacts. The environmental impacts associated with the entire life cycle were included in the group's proposed model (i.e., the group developed an extensive list of environmental impacts). The "costs" associated with these impacts were not evaluated.

A weighting method of cost estimation based on Colby's five environmental management paradigms was discussed. It was suggested that rather than deriving or assigning absolute weights, the weighting system could be used for sensitivity analysis over a range of values for the different impacts (Watkins, 1993). Colby's paper (1991) discussed the distinctions, connections, and implications for the future of environmental management by describing the changing strategies and the related philosophies of the following broad environmental management paradigms:

- frontier economics,
- environmental protection,
- resource management,
- eco-development, and
- deep ecology.

Associated with each paradigm are differing philosophies of human-nature relationships. The paradigms are overlapping and encompass several schools of thought. Colby's paper does not explicitly detail methods for evaluating environmental costs; however, it suggests that environmental costs would be treated differently according to the prevailing environmental management paradigm. The following is a description of the possible environmental costing methodologies under each of the five paradigms.

Frontier Economics

- Property owners and the public at large pay environmental costs (not necessarily the polluter).

- Production is limited by manmade factors. Natural factors are not accounted for. Analytic modeling and planning methodologies include net present value, maximization, and cost/benefit analysis of tangible goods and services.
- Economic analysis is based on the neoclassical model of the closed economic system.

Environmental Protection

- Taxpayers (public at large) pay environmental costs.
- Analytic modeling and planning methodologies include environmental impact assessment after design, optimum pollution levels, equation of WTP and compensation principles.
- Economic analysis is based on the neoclassical model of the closed economic system. Ecological benefits are difficult to quantify, so environmental management in this paradigm is treated strictly as an added cost.

Resource Management

- "Polluter" (producers and consumers) pays environmental costs.
- Analytic modeling and planning methodologies include natural capital; true (Hicksian) income; maximization of United Nations System of National Accounts; ecosystem and social health monitoring; and linkages between population, poverty, and environment.
- Economic analysis based on an extension of neoclassical economics that incorporates all types of capital and resources—biophysical, human, infrastructural, and monetary—into calculations of national accounts, productivity, and policies for development and investment planning.
- Pollution can be considered a "negative resource" (causing natural capital degradation), rather than an externality.
- The concern for nature stems from the fact that hurting nature is beginning to hurt economic man. Environmental expenditures are considered necessary to avoid "more" costs.

Eco-Development

- A "pollution prevention pays" concept rewards those that do not pollute. The economy is structured to reduce pollution as a throughput.
- Analytic modeling and planning methodologies include ecological economics; open system dynamics; socio-technical and ecosystem process design; integration of social, economic, and ecological criteria for technology; and trade and capital flow based on community goals and management.
- The relationship between society and nature can be considered a "positive sum game." Human activities are organized to be synergistic with ecosystem processes and services.

- Emphasis is placed on efficient, clean, renewable energy sources; environmental information; community consciousness; and experiential quality of economic activity.
- An example of the eco-development paradigm is the International Joint Commission between U.S. and Canada, which explicitly uses a stakeholder, positive-sum approach.

Deep Ecology

- Environmental costs avoided by foregoing development.
- Analytic modeling and planning methodologies include grassroots regional planning, multiple cultural systems, and conservation of cultural and biological diversity.

Strengths

One strength of the approach used by IMDI for assessing environmental impacts is that it provides a framework and methodology for breaking complex problems down into constituent parts. The method provides a framework for organizing complex issues into a more easily manageable format that defines goals, objectives, subobjectives, and criteria relating to environmental quality. The criteria may then be assessed individually against expert knowledge and stakeholder values to gain a better understanding of the problem at hand. Through the use of the IMDI methodology, coefficients can be established for various substances that indicate the relative environmental impact of those substances. Such coefficients may be directly compared, allowing for a relative comparison of individual substances or the evaluation of the environmental profile of an entire system.

Weaknesses

The primary weakness of the IMDI approach is that the process for developing weights for individual substances is highly subjective. It is not clear how weights developed by different groups could be compared against one another in a meaningful way. The AHP pair-wise comparison process is largely a subjective process requiring expert knowledge to rate the intensities of the environmental impacts being compared. In the case of most future problems, including potential environmental impacts, there is no such expert knowledge. Thus the weighting factors developed by different groups would not be very meaningful. In addition, the possibility exists that the experts can have a bias and/or misjudge the importance of particular attribute intensities. Because the IMDI approach relies on the values and judgment of a select group of individuals, the results from this approach probably could not be replicated.

The IMDI approach also concentrates solely on environmental quality and environmental impacts and thus is somewhat limited in its intended application to environmentally conscious manufacturing because additional factors (e.g., cost, functional requirements, performance) also contribute to decisions affecting product design (Watkins, 1993).

Relevance to Impact Assessment

The concept of assigning weights in the IMDI, particularly for a range of values, is particularly noteworthy to impact assessment because it attempts to provide a common metric for valuing environmental impacts. Because of the subjective nature of weighting process used in the IMDI approach, its use would be more appropriate for internal impact assessment applications. The IMDI approach requires further testing before results can be supported in external applications.

7.4 INTEGRATED SUBSTANCE CHAIN MANAGEMENT

Developed by VNCI (an association of the Dutch Chemical Industry), integrated substance chain management (also called the VNCI process) was designed to evaluate a substance throughout its entire life cycle (Canadian Standards Association, 1992). Integrated substance chain management was also designed to encourage the use of environmentally preferable substitutes and recycling alternatives and the identification and closure of leaks.

To include all environmental issues, each link in the substance chain is checked against a comprehensive list of environmental themes, including global warming, ozone depletion, acidification, eutrophication, photochemical ozone formation, dispersion of toxic substances, disposal of wastes, and disruption/depletion of natural resources.

Based on rough estimates of product system inputs and outputs and their associated environmental issues, options for process improvement can be proposed. The selection of options for detailed analysis is based on

- environmental impact,
- cost effectiveness, and
- relevance to decisionmakers (Canadian Standards Association, 1992).

The output of the detailed analysis is a two-axis (environmental impact/economic impact) options map. The options map is developed by determining the environmental and economic profiles of the substance in question and positioning the various options (see Figure 7-3).

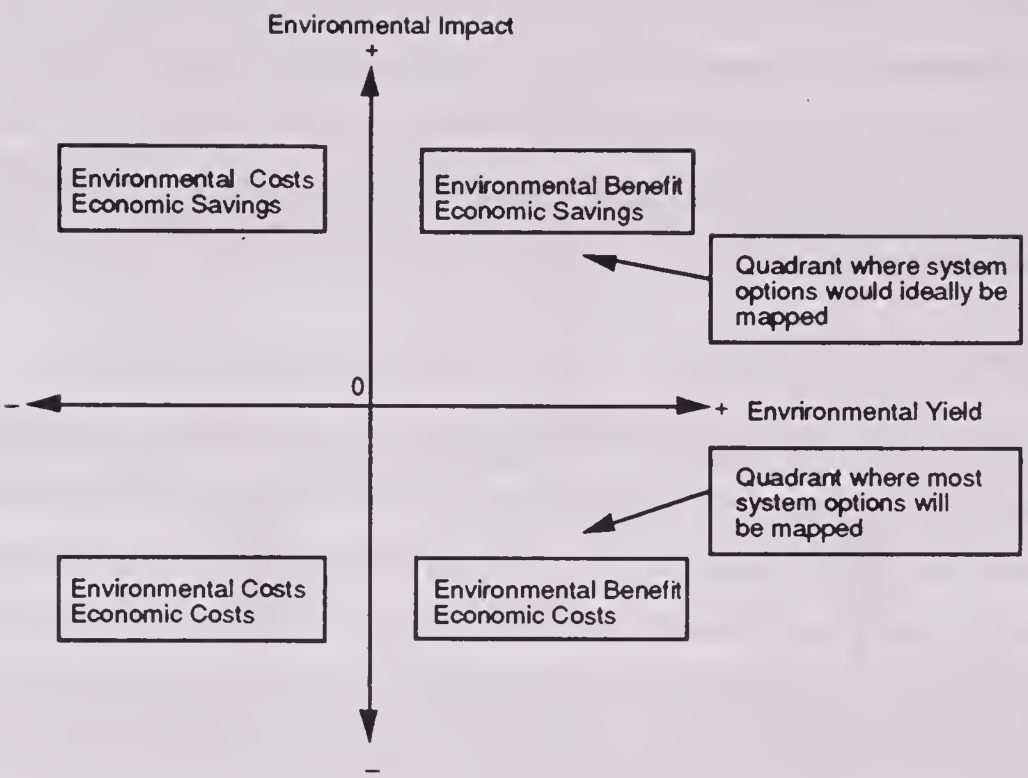

Figure 7-3. Options Map for Integrated Substance Chain Management

Source: Canadian Standards Association, 1992

The environmental profile provides a comprehensive overview of the relevant environmental impacts associated with each process option. Impacts are quantified in terms of a single unit of measurement for each impact category (e.g., tonnes of CO_2 equivalent as an agent of global warming) and shown in terms of a fraction of the total national emission of that environmental impact theme. Exact changes of inputs and outputs associated with each process option are calculated and a checklist is used to determine the extent of changes in other input/output factors. The data are then converted to scales for measuring the impact on each environmental theme, and sensitivity analysis examines how changes in the underlying database of inputs, outputs, and conversion factors affect the results of the analysis.

The economic profile evaluates the economic impact of the proposed process options. When the environmental and economic profiles are completed, all quantitative figures and qualitative comments in each profile are combined to arrive at a final conclusion concerning the total environmental and economic impacts.

Relative weights are then assigned to each environmental theme to enable the aggregation of environmental impacts associated with each process option. The environmental and economic impacts are then combined to represent a single point on the options map. The origin of theoptions map represents the "do nothing" option. Options that represent both environmental and economic improvements will be plotted in the upper right-hand quadrant. Options that represent both environmental and economic setbacks will be plotted in the lower left-hand quadrant.

Strengths

The main strength of the integrated substance chain management approach is that it provides a framework and methodology for integrating environmental concerns, economic concerns, and stakeholder values. The resulting options map portrays the environmental and economic differences between process options, allowing for relatively easy and objective decisionmaking. In addition, the development of relative weights for each environmental theme enables the environmental impacts associated with each process option to be aggregated to yield an overall environmental profile for the system.

Weaknesses

The main weakness of the integrated substance chain management approach is that it employs a relatively simplistic weighting scheme and may not be applicable to in-depth assessments of impacts. Additionally, it is not clear how life-cycle economic costs would be developed for use in the options map.

Relevance to Impact Assessment

The integrated substance chain management approach would be most applicable to internal impact assessments where a number of different factors (e.g., environmental protection, economic well being, public image) affect the environmental decisionmaking process and a less detailed level of assessment provides adequate information to make the decisions at hand.

7.5 ECO-RATIONAL PATH METHOD (EPM)

EPM represents a procedure that builds on the ESR method described in Chapter 4 to integrate environmental and economic information—two of the primary dimensions of environmental decisionmaking. The process for integrating these two dimension comprises three main steps—recording, judgment, and decision—as shown in Figure 7-4.

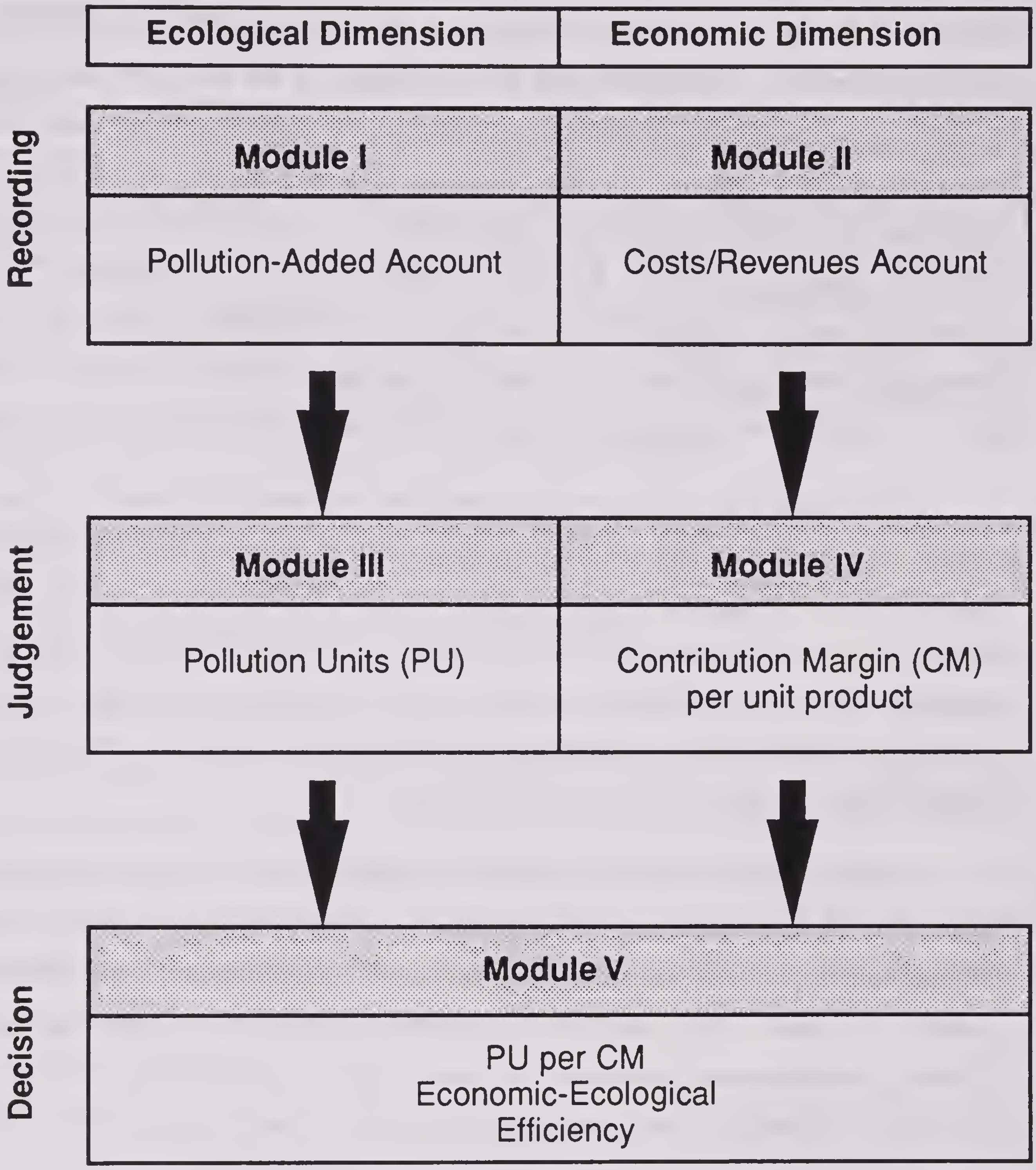

Figure 7-4. Conceptual Framework for the EPM

Source: Schaltegger and Sturm, 1993

Looking at the ecological dimension in Figure 7-4, the first step is to collect and record information on environmental releases. Releases in the context of EPM include inputs, desired output, and undesirable outputs. Although labeled "pollution-added account" (see Module I in Figure 7-4) the information for this step may be generated through using traditional inventory analysis procedures. Sometimes the data developed in the inventory analysis are sufficient for evaluating environmental improvement options, but often inventory data alone are insufficient. For example, when one system releases more CO_2 and another system releases more NO_x, then no obvious and objective judgments are possible. To weigh one pollutant against another, a preference ranking is needed. This step, termed judgment, is a procedure for developing weights or "pollution units" (see Module III in Figure 7-4) for releases according to their environmental relevance (based on ambient concentration standards for various media). Developing these relative weights is accomplished using the ESR method as described in Chapter 4.

With regards to the economic dimension as shown in Figure 7-4, the first step (see Module II) is to collect and record information on economic costs/revenues including environmental compliance costs and earnings. This information is typically generated in traditional accounting practices but may need to be broken out of an aggregate account (e.g., overhead) and appropriately allocated to a specific product or process. After all the necessary cost/revenue information is recorded, the contribution margin (see Module IV in Figure 7-4) is calculated as a measure of economic efficiency.

The integration of the economic and ecological information is shown in Module V in Figure 7-4. In this module, the data produced from Modules III and IV are integrated by calculating the quotient pollution units per, for example, created dollar contribution margin of a product or process. This calculation provides a measure of the economic-ecological efficiency of specific products and processes. In general, the most preferable products or processes are those with low pollution units and high contribution margin (i.e., small PU/CM ratio).

Strengths

Some of the strengths of the EPM include the following:

- framework and methodology are provided for integrating environmental and economic considerations;
- weighting factors using ambient regulatory standards represent social, political, regulatory, and scientific opinions and values;

- weighting schemes used in EPM represent the relative environmental and economic impacts of different chemical releases to different environmental media;
- EPM is flexible and can be used at a variety of different spatial levels (e.g., state, regional, and local).

Weaknesses

One weakness of the EPM is that the ESR weighting scheme's use of relations between ambient standards is not a natural scientific or ecotoxicological based scheme, but instead represents a socio-cultural judgment from an ecological perspective (which relies on ecotoxicological data). However, no objective and undoubtedly valid opinion on the harmfulness of substances exists. ESR develops weights according to generally accepted norms and values, which are theoretically expressed in ambient concentration standards. Such ambient standards may or may not reflect actual environmental impacts.

Another weakness of the EPM is that the method for characterizing the economic impacts of pollutants is somewhat simplistic. It is not clear whether the results of the economic impact assessment component would be useful to decisionmakers.

Relevance to Impact Assessment

The information produced from applying the EPM can be used in impact assessment to evaluate and compare the relative environmental and economic impacts of inventory items where regulatory standards exist. Although using the ESR approach provides a consistent estimate of the environmental impacts, it does not necessarily preclude the need for additional analyses. EPM requires additional testing in the context of LCA to better gauge its applicability to impact assessment.

CHAPTER 8

KEY POINTS AND FUTURE RESEARCH NEEDS

The purpose of life-cycle impact assessment is to translate the results of an inventory analysis into a description of environmental impacts, providing users with additional information to discern between alternatives (e.g., inventory items, systems). Impact assessment also makes explicit the methods used to compare and weigh alternatives. This document covers a wide variety of issues related to impact assessment and outlines existing methods that have been used or presented in the context of impact assessment. Again, it should be kept in mind that this document is not a guidance document, but rather a compendium on the state of practice of impact assessment.

This final chapter summarizes some of the key points discussed throughout this document and provides a listing of potential research needs for the future research and development of impact assessment techniques and methods. While impact assessment is still in its infancy, this document illustrates promise for the current applications and future development of impact assessment techniques and methods.

8.1 SUMMARY OF KEY POINTS

This document covers a broad range of material which cuts across a variety of research areas. Some of the key points that can be drawn are summarized below.

- Impact assessment has been conceptually defined to include three phases: classification of inventory items into impact categories, characterization of potential impacts, and valuation of impacts. However, formal procedures and methods for conducting impact assessment have not yet been established.
- Impact assessment may be useful for a variety of both internal and external applications (see Chapter 1). Although internal applications may not be required to follow stringent LCA guidelines, they should nonetheless follow the best practice.
- Practitioners may not need to complete a full impact assessment to obtain useful information. In some cases, merely classifying inventory items into impact categories may provide adequate information for users to identify improvement options. In other cases, a more detailed impact assessment information may be needed.
- A wide variety of methods are available for use in impact assessment (see Chapters 4 through 7), ranging from simplistic checklists to complete risk and economic impact assessments. A rule-of-thumb for choosing the appropriate method(s) is to choose the method(s) that provides adequately detailed information to make the decision at hand

(usually to discern the relative impact of different substances). A more complex method(s) is used only when the resulting information is needed to advance the decision to be made.

- There is a general lack of methods for assessing the impacts of nonchemical loadings (e.g., habitat alteration, heat, noise) to the environment. Many of the existing methods for characterizing impacts (see Chapter 4) are based upon chemical exposure and toxicology data and cannot readily be used to assess the impact of nonchemical loadings.
- There are a number of places in the impact assessment where value judgments may play a significant role. It is critical that practitioners document points in the impact assessment process where value judgments were employed, the set of values used to make the judgment, and how those judgments may affect the outcome of the impact assessment;
- A significant level of uncertainty is associated with impact assessment (e.g., linking inventory items to impacts). Uncertainty, however, is a fact of life for virtually all areas of research. Although there are currently no formal procedures for evaluating uncertainty in impact assessment, practitioners should nonetheless document and evaluated sources of uncertainty and appropriately qualify impact assessment results.
- As with other LCA components, it is critical that practitioners clearly communicate the content and conduct aspects of the impact assessment in the final LCA report. This includes, but is not limited to, the goals and scope, data sources used and their quality, models used and their assumptions and limitations; and data or methodology manipulations; value judgments employed, and the analyst's interpretation of these aspects on the overall LCA results.

8.2 POTENTIAL FUTURE RESEARCH NEEDS

Potential research needs (adapted from Vigon and Evers, 1992) identified by the LCA community regarding the future development and application of impact assessment tools and procedures are listed in Table 8-1.

TABLE 8-1. POTENTIAL FUTURE NEEDS FOR IMPACT ASSESSMENT RESEARCH

Research Needs	Effort
• Relate EIS scoping process to impact assessment	Low
• Define impact descriptors for LCA applications	Low
• Determine basis for defining stock resource pool	Low
• Develop impact category equivalency factors	Moderate
• Identify ecohazard profile parameters, thresholds	Moderate
• Develop method for evaluating depletion of water resources	Moderate
• Develop method for linking resource development with depletion	Moderate
• Achieve international consensus on impact assessment	Moderate
• Assess feasibility of nonchemical impacts matrix	Moderate
• Prepare impact analysis technical support document	Moderate
• Develop and validate of streamlined impact assessment methods	Moderate
• Develop a reference data base of generic impact assessment information	High
• Develop library of impact networks	High
• Prepare broad range of impact assessment case studies	High
• Develop methods for factoring uncertainty into impact assessments	High
• Determine feasibility of resource management/economic models for LCA	High
• Develop methods for estimating biodiversity change and habitat alteration	High
• Develop models to assess susceptibility due to health stress	High
• Develop better human exposure models within an LCA	High
• Evaluate ecological risk assessment models/methods	High
• Develop/validate ecological hazard matrix approach	High
• Fill data gaps in the following areas: — Health exposures — Short-term and long-term bioassays — Effects of unintended product use — Exposure from nonmanufacturing — Nonpoint sources of pollution	High

Source: Modified from Vigon and Evers, 1992

Appendix A
National Environmental Policy Act (NEPA) Environmental Assessment Procedures

Under Section 102 of the National Environmental Policy Act (NEPA), federal agencies are required to make a full and adequate analysis of all environmental effects of implementing its programs or actions (Jain et al., 1993). In the context of NEPA, an environmental impact assessment (EIA) is used for determining if a more detailed environmental impact statement (EIS) is required. EIAs utilize a list of environmental "attributes" for which baseline values are compared against actual or expected values to determine the level of potential impact. After the environmental "attributes" are determined, the EIA scoping process is used to evaluate and streamline a comprehensive list of "attributes" or impacts.

The comprehensive list of environmental attributes considered in an EIA and the scoping process used to streamline that comprehensive list to a reference project may be useful in the context of impact assessment where a wide variety of impacts require consideration. These components of EIA are described in further detail in the following sections.

A.1 ENVIRONMENTAL ATTRIBUTES ADDRESSED IN EIA

Environmental attributes are variables that represent characteristics of the environment (see Table A-1). The environment is difficult to characterize because it contains numerous attributes exhibiting complex interrelationships. However, anticipated changes in the attributes of the environment and their interrelationships are defined as potential impacts. All lists of environmental attributes are a shorthand method for focusing on important characteristics of the environment. Because of the complex nature of the environment, any such listing is limited and, consequently, may not capture every potential impact. The more complete the listing is, the more likely it will reflect all important effects on the environment, but this list may be expensive and cumbersome to apply.

Table A-2 summarizes possible environmental attributes in eight categories that comprise the biophysical and socioeconomic environment at a generalized level. While this list of attributes represents a reasonable breakdown of environmental parameters, it is likely to require modification or supplementation depending on the type of action to be assessed. For a more complete description of these attributes, the reader is referred to Jain et al. (1993).

A.2 EIA SCOPING PROCESS

When EIAs were first introduced, decisionmaking based on EISs was being compromised by their inclusion of what many considered to be insignificant factors. These insignificant factors were considered to be background noise, while significant factors were in

TABLE A-1. ENVIRONMENTAL ATTRIBUTE CATEGORIES USED IN EIA

Environmental Attributes	
Air	**Ecology**
• Diffusion factor	• Large animals (wild and domestic)
• Particulates	• Predatory birds
• Sulfur oxides	• Small game
• Hydrocarbons	• Fish, shellfish, and waterfowl
• Nitrogen oxide	• Field crops
• Carbon monoxide	• Threatened species
• Photochemical oxidants	• Natural land vegetation
• Hazardous toxicants	• Aquatic plants
• Odors	
	Sound
Water	• Physical effects
• Aquifer safe yield	• Psychological effects
• Flow variation	• Communication effects
• Oil	• Performance effects
• Radioactivity	• Social behavior
• Suspended solids	
• Thermal pollution	**Human Aspects**
• Acid and alkali	• Lifestyles
• Biochemical oxygen demand (BOD)	• Psychological needs
• Dissolved oxygen (DO)	• Physiological systems
• Dissolved solids	• Community needs
• Nutrients	
• Toxic compounds	**Economics**
• Aquatic life	• Regional economic stability
• Fecal coliforms	• Public-sector review
	• Per capita consumption
Land	
• Soil stability	**Resources**
• Natural hazard	• Fuel resources
• Land-use patterns	• Nonfuel resources
	• Aesthetics

Source: Jain et al., 1993.

danger of being concealed and possibly overlooked. "Scoping" was introduced in EIA as a process used to determine the range (i.e., scope) of issues to be addressed. CEQ regulations require using the scoping process early in the planning stages, as soon as practicable after agency decision to prepare an EIS.

TABLE A-2. ENVIRONMENTAL ATTRIBUTES USED IN EIA

Attribute	Variables to be Measured	Data Sources	Mitigation of Impact
Air			
Diffusion Factor	• Stability • Mixing depth • Wind speed • Precipitation • Topography	Primary sources of data are the National Weather Service and the United States Geological Survey (USGS).	Mitigation techniques have not been adequately defined.
Particulates	The concentration of all solid and liquid particles averaged annual arithmetic mean of all 24 h particulate concentrations at a given location.	Data sources include state pollution control departments, county air pollution control offices, multi-county air pollution control offices, or city air pollution control offices.	• Source reduction • Reduction or removal of receptors from the area • Particulate removal devices • Use of protected, controlled environments
Sulfur Oxides	The 24 h annual arithmetic mean concentration of SO_2 present in the ambient air.	Data are generally compiled and published annually by air quality monitoring programs established by state pollution control agencies; the EPA; and county, regional, multi-county, or city air pollution control agencies.	• Source reduction • Reduction or removal of receptors from polluted areas • Gas removal devices using absorption, adsorption, and catalytic converters • Use of protected, controlled environments
Hydrocarbons	The 3 h average annual concentration of ambient hydrocarbons, expressed in ppm, and measured between 6 and 9 a.m. (peak hydrocarbon concentration time).	Data are generally available from state air quality monitoring programs. Other potential sources include the EPA and city or county monitoring agencies.	• Control of motor vehicle emissions • Control of stationary source emissions • Reduction or removal of receptors from area • Use of a controlled environment

(continued)

TABLE A-2. ENVIRONMENTAL ATTRIBUTES USED IN EIA (CONTINUED)

Attribute	Variables to be Measured	Data Sources	Mitigation of Impact
Air (continued)			
Nitrogen Oxide	The average annual concentration of nitrogen oxides in the ambient air, measured in ppm.	Sources of data include state pollution control departments and county, multi-county, or city air pollution control offices.	• Control of motor vehicle emissions • Control of stationary source emissions • Reduction or removal of receptors from area • Gas removal devices using absorption, adsorption, and catalytic converters • Use of a controlled environment
Carbon Monoxide	The maximum 8 h and 1 h concentration of carbon monoxide measured in micrograms per cubic meter.	Sources of data include the state pollution control department, the county air pollution control office, or the city air pollution control office.	• Control of motor vehicle emissions • Control of stationary source emissions • Reduction or removal of receptors from area
Photochemical Oxidants	The maximum hourly average concentration measured in micrograms per cubic meter.	Sources of data include the state pollution control department, the county air pollution control office, or the city air pollution control office.	• Control of motor vehicle emissions • Control of stationary source emissions • Reduction or removal of receptors from area • Gas removal devices using absorption, adsorption, and catalytic converters • Use of a controlled environment

(continued)

TABLE A-2. ENVIRONMENTAL ATTRIBUTES USED IN EIA (CONTINUED)

Attribute	Variables to be Measured	Data Sources	Mitigation of Impact
Air (continued)			
Hazardous Toxicants	The variable to be measured varies with the toxicant.	Only a few city, county, regional, and state agencies monitor hazardous toxicants and emissions. Data on toxicant monitoring are available from state and local air pollution control agencies when collected.	• Use of materials that do not generate hazardous toxicants • Use of processes that do not generate hazardous toxicants • Source reduction • Control, removal devices • Moving people from contaminated areas
Odors	• The average annual concentration of selected odor contaminants in ppm by volume. • The odor intensity, rated from 0 (no odor) to 4 (strong odor) by a panel.	No systematic monitoring and data collection are done by state and local agencies.	• Dilution of odorant • Odor counteraction • Odor masking • Source reduction • Removal or receptors from polluted areas, and/or downwind odor path fatigued olfactory odor perception
Water			
Aquifer Safe Yield	The amount of water withdrawn in a unit of time, usually expressed as thousands of acre-feet of water per annum.	Sources of data include local USGS offices and state water agencies.	All activities likely to change the physical nature of the aquifer, land surface runoff, and percolation. Water availability to the aquifer should be carefully controlled.
Flow Variations	• The typical unit of flow measurement is cubic feet per second. • Velocity as measured in feet per second.	Data sources include local Army Corps of Engineers offices and state water agencies.	All activities such as land use projects and water impoundment and operation should be considered minimize flow variations from the mean natural flow.

(continued)

TABLE A-2. ENVIRONMENTAL ATTRIBUTES USED IN EIA (CONTINUED)

Attribute	Variables to be Measured	Data Sources	Mitigation of Impact
Water (continued)			
Oil	Quantitative: • milligrams of oil or grease per liter of water Qualitative: • visible oil slick • oily taste/odor • coating of banks or bottom	Data sources include local Army Corps of Engineers offices and state water agencies.	• Controlling all direct discharge • Treatment of surface runoff for oil separation • Restrict lagooning of oil wastes to prevent potential groundwater contamination
Radioactivity	• The quantity of any radioactive material in which the disintegrations per second are 3.7×10^{10}, expressed as Curie (Ci) • Microcurie (10^{-6}Ci) • Picocurie (10^{-12}Ci)	Data may be obtained from the Nuclear Regulatory Commission (NRC) and state water agencies.	• Waste containing radioactivity should be treated separately by means of dewatering • Monitoring and control of radiation facilities
Suspended Solids	• Readily settleable suspended solids are measured in milliliters per liter of settled water.	Sources of data include local USGS offices, local Army Corps of Engineers offices, and state water agencies.	• Controlling/treatment of discharge, including sanitary sewage and industrial wastes • Minimize activity that increases erosion or contributes nutrients to water
Thermal Pollution	Water temperature measured in degrees Centigrade or Fahrenheit.	Sources of data include local USGS offices, local Army Corps of Engineers offices, and state water agencies.	Use of cooling towers in a closed-loop water cooling system.
Acid And Alkali	pH	Sources of data include local USGS offices, local Army Corps of Engineers offices, and state water agencies.	• Neutralization of acidic or alkaline waters by incorporation of alkaline or acid wastes, respectively • Source reduction of acid or alkaline wastes

(continued)

TABLE A-2. ENVIRONMENTAL ATTRIBUTES USED IN EIA (CONTINUED)

Attribute	Variables to be Measured	Data Sources	Mitigation of Impact
Water (continued)			
Biochemical Oxygen Demand (BOD)	The amount of oxygen consumed (mg/L) by organisms during a five-day period at 20°C.	Sources of data include local USGS offices, local Army Corps of Engineers offices, and state water agencies.	• Treatment of all wastes containing organic material: - biological - chemical - packaged units
Dissolved Oxygen (DO)	Milligrams of oxygen per liter of water.	Sources of data include local USGS offices, local Army Corps of Engineers offices, and state water agencies.	• Treatment of all wastes containing organic material: - biological - chemical - packaged units
Dissolved Solids	Total dissolved solids, determined after evaporation of a sample of water and its subsequent drying at 103°C.	Sources of data include local USGS offices, local Army Corps of Engineers offices, and state water agencies.	• Controlled landfilling to avoid possible leaching • Deep well injection of brine • Control and treatment of surface runoffs
Nutrients	Includes measurement of phosphorus, nitrogen, carbon, iron, trace metals in their appropriate terms.	Sources of data include local USGS offices, local Army Corps of Engineers offices, and state water agencies.	• Waste water treatment • Natural assimilation
Toxic Compounds	The spectrum of toxic materials is extremely large and highly diverse in terms of effects. Measurement can be expressed as µg/L for specific compounds.	Sources of data include local USGS offices, local Army Corps of Engineers offices, and state water agencies.	• Monitor and control of all toxic wastes • Dilution
Aquatic Life	Field observations	Data may be obtained from local Fish and Wildlife offices.	Control and reduction of all water quality attributes listed

(continued)

TABLE A-2. ENVIRONMENTAL ATTRIBUTES USED IN EIA (CONTINUED)

Attribute	Variables to be Measured	Data Sources	Mitigation of Impact
Water (continued)			
Fecal Coliforms	Coliform density, reported in terms of coliform per 100 mL.	Sources of data include local Army corps of Engineers offices and state water agencies.	• Treatment of all wastes containing organic material: - biological - chemical - packaged units
Land			
Soil Stability (Erosion)	• Soil composition • Degree of slope • Length of slope • Nature and extent of vegetative cover • Intensity/frequency of exposure to eroding forces	Data are generally available from local U.S. Soil Conservation Service offices.	• Erosion control devices: - ground cover - tile drainage - grassed waterways - terracing steep slopes - catch basins
Natural Hazard	Specific to each type of hazard.	Sources of data include the Corps of Engineers, USGS, U.S. Forest Service, National Weather Service, state geologists, and local universities.	Specific to each type of hazard.
Land-Use Patterns	Compatibility of use between parcels as indicated by such variables as: • type and intensity of use • noise • transportation pattern • prevailing wind direction • buffer zones • aesthetics	Municipal land use plans, county land use planning commission, regional land use council, Bureau of Land Management, National Park Service, Bureau of Reclamation, Corps of Engineers, Tennessee Valley Authority, and the Department of Energy.	• Inclusion of buffer zones • Use of zoning and land use ordinances • Community participation in the land use planning process

(continued)

TABLE A-2. ENVIRONMENTAL ATTRIBUTES USED IN EIA (CONTINUED)

Attribute	Variables to be Measured	Data Sources	Mitigation of Impact
Ecology			
Large Animals (Wild and Domestic)	• Population • Number of species • Habitat (in hectares) • Human intrusion/noise	Data sources include the U.S. Fish and Wildlife Service, wildlife experts, and universities.	• Minimize human intrusion/noise • Creation of National Parks, National Wildlife Areas, or other protected areas of habitat
Predatory Birds	• Population • Number of species • Habitat (in hectares) • Human intrusion/noise	Data sources include the U.S. Fish and Wildlife Service, wildlife experts, and universities.	• Minimize human intrusion/noise • Creation of National Parks, National Wildlife Areas, or other protected areas of habitat
Small Game	• Population • Number of species • Habitat (in hectares) • Human intrusion/noise	Data sources include the U.S. Fish and Wildlife Service, wildlife experts, and universities.	• Minimize human intrusion/noise • Creation of National Parks, National Wildlife Areas, or other protected areas of habitat
Fish, Shellfish, And Waterfowl	• Population • Number of species • Habitat (in hectares) • Human intrusion • pH • BOD • DO • Coliform bacteria • Pesticide concentrations	Data sources include the U.S. Fish and Wildlife Service, wildlife experts, and universities.	• Minimize human intrusion/noise • Creation of National Parks, National Wildlife Areas, or other protected areas of habitat
Field Crops	• Acres of land • Percent farmed • Type of crop • Natural habitat (in hectares) • Human intrusion		• Minimize human intrusion/noise • Minimize use of pesticides and herbicides

(continued)

TABLE A-2. ENVIRONMENTAL ATTRIBUTES USED IN EIA (CONTINUED)

Attribute	Variables to be Measured	Data Sources	Mitigation of Impact
Ecology (continued)			
Threatened Species	• Population • Number of species • Habitat (in hectares) • Human intrusion	Data sources include the U.S. Fish and Wildlife Service, wildlife experts, and universities.	• Minimize human intrusion/noise • Creation of National Parks, National Wildlife Areas, or other protected areas of habitat • Breeding programs
Natural Land Vegetation	• Acres of native vegetation • Number and types of species • Human intrusion	Data sources include the U.S. Fish and Wildlife Service, wildlife experts, and universities.	• Minimize land conversion • Restrict vehicular intrusion • Creation of National Parks, National Wildlife Areas, or other protected areas of habitat
Aquatic Plants	• Population • Number of species • Habitat (in hectares) • Human intrusion • pH • BOD • DO • Coliform bacteria • Pesticide concentrations	Data sources include the U.S. Fish and Wildlife Service, wildlife experts, and universities.	• Minimize waste and nutrient inputs • Restrict drainage of wetlands • Creation of National Parks, National Wildlife Areas, or other protected areas of habitat

(continued)

TABLE A-2. ENVIRONMENTAL ATTRIBUTES USED IN EIA (CONTINUED)

Attribute	Variables to be Measured	Data Sources	Mitigation of Impact
Sound			
Physical Effects	• Loudness, measured in decibels (dB) • Duration • Frequency	Under the Noise Control Act of 1972, EPA promulgates noise-emission standards for construction and transportation equipment, motors/engines, and electrical equipment. Data for construction noise are provided by the General Services Administration. OSHA provides noise exposure criteria for occupational health.	• Source reduction • Dampening • Dissipation • Deflection • Ear protection • Sound enclosures • Removal of receptors from high noise areas
Psychological Effects	• Loudness, measured in decibels (dB) • Duration • Frequency • Psychological stress	Under the Noise Control Act of 1972, EPA promulgates noise-emission standards for construction and transportation equipment, motors/engines, and electrical equipment. Data for construction noise are provided by the General Services Administration. OSHA provides noise exposure criteria for occupational health.	• Source reduction • Dampening • Dissipation • Deflection • Ear protection • Sound enclosures • Removal of receptors from high noise areas
Communication Effects	• Loudness, measured in decibels (dB) • Duration • Frequency • Ambient noise levels • Distance between speaker and listener	Under the Noise Control Act of 1972, EPA promulgates noise-emission standards for construction and transportation equipment, motors/engines, and electrical equipment. Data for construction noise are provided by the General Services Administration. OSHA provides noise exposure criteria for occupational health.	• Source reduction • Dampening • Dissipation • Deflection • Ear protection • Sound enclosures • Removal of receptors from high noise areas • Use of headsets

(continued)

TABLE A-2. ENVIRONMENTAL ATTRIBUTES USED IN EIA (CONTINUED)

Attribute	Variables to be Measured	Data Sources	Mitigation of Impact
Sound (continued)			
Performance Effects	Loudness, measured in decibels (dB)	Under the Noise Control Act of 1972, EPA promulgates noise-emission standards for construction and transportation equipment, motors/engines, and electrical equipment. Data for construction noise are provided by the General Services Administration. OSHA provides noise exposure criteria for occupational health.	• Source reduction • Dampening • Dissipation • Deflection • Ear protection • Sound enclosures
Social Behavior Effects	• Loudness, measured in decibels (dB) • Duration • Frequency • Ambient noise levels • Distance between speaker and listener	Under the Noise Control Act of 1972, EPA promulgates noise-emission standards for construction and transportation equipment, motors/engines, and electrical equipment. Data for construction noise are provided by the General Services Administration. OSHA provides noise exposure criteria for occupational health.	• Source reduction • Dampening • Dissipation • Deflection • Ear protection • Sound enclosures • Removal of receptors from high noise areas
Human Aspects			
Lifestyles	Variables to be measured for this attribute cannot be precisely defined. The objective is to identify general changes in social activities that will be caused by the proposed action.	Data for this attribute may be generally obtained from the predictions by community social leaders, local political leaders, academics, etc.	Although impact to this attribute cannot be completely mitigated, the effect of anticipated impacts could be lessened by forewarning participants.

(continued)

TABLE A-2. ENVIRONMENTAL ATTRIBUTES USED IN EIA (CONTINUED)

Attribute	Variables to be Measured	Data Sources	Mitigation of Impact
Human Aspects (continued)			
Psychological Needs	Although no specific variables are identified for this attribute, a general feeling of the degree to which the psychological needs of individuals and communities are being met can be obtained.	Data on this attribute can be generally obtained from psychologists, personal surveys, local counselors, clergy, and law enforcement officials.	• Including an action plan that would provide assistance for affected individuals • Consultation • Social programs
Physiological Systems	No variables can be measured for this attribute. The detailed activities and implications of those activities must be carefully examined.	Data on this attribute can be generally obtained from psychologists, personal surveys, local counselors, clergy, and law enforcement officials.	• Taking precautionary measures to avoid the impact. • Employing specific safety practices • Using protective devices
Community Needs	• Population • Demographics • Available housing • Capacity of public services • Characteristics of land use	Data may be obtained from public surveys, local planning agencies, police and fire departments, local officials.	Including a plan for providing public services to accompany the proposed activity.
Economics			
Regional Economic Stability	Percentage of total regional economic activity affected.	Data sources include local and regional business and employment statistics.	• Increase the demand for the output of highest growth industries in the region • Change the distribution of demand for the output

(continued)

TABLE A-2. ENVIRONMENTAL ATTRIBUTES USED IN EIA (CONTINUED)

Attribute	Variables to be Measured	Data Sources	Mitigation of Impact
Economics (continued)			
Public Sector Review	• Annual average revenues and expenditures of the relevant government agencies • Expenditures necessary to provide adequate public services without the project	Data sources include State and Local Finances, and the Statistical Abstracts of the United States.	Design project activities to either reduce social costs or increase payments to the local government.
Per Capita Consumption	Average amount that will be spent in each future year throughout the life of the project by affected individuals.	Data may be obtained from State and Local Finances.	Establish direct linkages with area industries, businesses, or other economic activities to encourage inflows of money.
Resources			
Fuel Resources	• Rate of fuel consumption (in Btu) • Useful energy output derived from fuel consumption • Heat content of fuels • Types of fuel	Data sources includes the Gas Engineer's Handbook Mining Statistics, Energy Information Administration, and State and Local Statistics	• Alternate fuel selection • Conservation of fuel resources
Nonfuel Resources	• Points of resource consumption • Consumption rates • Quantities and content of wastes from resource acquisition activities	Data sources include the Gas Engineer's Handbook Mining Statistics, Energy Information Administration, and State and Local Statistics.	• Economizing on resource requirements • Development and use of substitutes • Recycling programs
Aesthetics	Individual perception and values for defining beauty make it difficult to quantify aesthetic impacts.	Data may be obtained from surveys, and other specific measurements.	• Public participation in planning processes • Designation of natural areas

Source: Jain et al., 1993.

In the first part of the EIA scoping process, a comprehensive list of impacts is streamlined to a particular study to minimize the proliferation of insignificant items (Jain et al., 1993). Impacts cannot be eliminated from this comprehensive list without first evaluating the significance or relevance of those impacts to the proposed project. For example, it would be inappropriate for a proposed project to consider impacts to a timber resources category if the project does not utilize, or produce an adverse impact on, timber resources. Thus, not only does this scoping process reduce inefficient use of time and resources, but it also helps to pinpoint the most critical impacts for analysts and decisionmakers to consider.

The second part of the EIA scoping process entails the *tiering* of impacts. Tiering comes into play when some of the impact categories on the "long list" are potentially affected by a project, but they are of fairly insignificant consequence. Such impacts are tiered to a lower level of importance and not initially evaluated in the study (although they may be evaluated during the study if necessary). The EIAs used tiering to organize the comprehensive list of impacts in a more manageable and meaningful manner, by differentiating relatively insignificant and significant impacts. The same problem may exist in impact assessment where a practitioner may need to evaluate potentially large numbers of impacts in the classification phase and streamline the list in the characterization phase.

From these two activities, a comprehensive list of environmental impacts may be tailored to a specific reference project to help analysts and decisionmakers pinpoint and address the critical impacts associated with the project.

In summary, the EIA scoping process requires an early analysis of potential impacts with reference to a specific project. The scoping process strives to

1) eliminate inappropriate impact categories from the analysis,
2) tier less important impact categories to a lower level of analysis, and
3) identify the critical impacts that must be addressed in the analysis.

Appendix B
Additional Impact Assessment Methods

This appendix contains descriptions of methods that have been evaluated for applicability to impact assessment but have not been tested or presented in the context of LCA. As in the presentation of methods in Chapters 4 through 7, the methods in this Appendix are presented in the order of increasing level of detail.

B.1 GREEN INDICATORS

Green indicators are calculated characteristics of a product or process that may be used to evaluate the environmental compatibility of the product or process by identifying indicators that are undesirably high or low. The ultimate goal of the green indicators is to give environmental concerns equal weight with other more traditional concerns, such as manufacturing and reliability, as part of an overall approach to green engineering design (Navinchandra, 1991). Table B-1 lists some green indicators that may be useful for a simple impact assessment (i.e., loading-type assessment).

Strengths

The primary strength of using the green indicators is that they provide a multi-dimensional view of a product system that can enable decisionmakers to simultaneously address a wide variety of issues and concerns. For example, most environmental assessment techniques only provide information on environmental effects. Green indicators provide information not only on environmental effects, but also on product performance, recyclability, useful life, cost, etc. Such information is integral to making high quality decisions concerning tradeoffs between alternatives products, processes, and materials as well as between environmental, economic, and production concerns.

Some additional strengths of the green indicators include the following:

- relatively convenient and easy to calculate,
- limited amount of external data is required,
- can be used as part of an overall green product design program, and
- involves a life-cycle perspective.

Weaknesses

One possible weakness of the green indicators is that they do not estimate environmental impacts per se. Rather, the indicators are merely proxies that can be related to environmental impacts. For example, although the degradability indicator provides an estimate of the portion of material in a product that is degradable, it does not indicate how harmful the degradable and

TABLE B-1. EXAMPLE GREEN INDICATORS

Indicator	Description
Percent Recycled	The percentage of recycled material in a product.
Degradability	The ratio of the volume of degradable material in a product to the total volume of the product.
Life	The time it takes for the degradable portion of a product to degrade. A curve showing the expected volume of reduction over time is used to determine life.
Junk Value	This is a measure of the total time a product will take to degrade into the environment. It is calculated as the area under the life curve (above) and expressed in units of cubic inches per year.
Separability	A measure of what materials can be separated from a product. It is the ratio of the volume of separable materials to the total volume of the product. (The notion of separability is different from disassembly.)
Potential Recyclability	The ratio of the volume of recyclable materials to that of unrecyclable materials.
Possible Recyclability	Composites and glued materials are potentially recyclable but cannot be recycled because they are inseparable. This indicator must be measured on a part-by-part basis and must take into account the available recycling methods and their economic viabilities.
Useful Life	When a material leaves the environment and enters the human world it is being used. Useful life is defined as the time an item spends in the activity for which it was designed.
Utilization	The ratio of the useful life of a product or material to the time it takes to "return" to the environment.
Net Emissions	The respective sums of solid, gaseous, and waterborne emissions from a particular product or process life cycle.
Total Emissions	The sum of all solid, gaseous, and waterborne emissions taken together from a particular product or process life cycle.
Total Hazardous Fugitives	A measure of the weight of hazardous fugitives, expressed as the ratio of the weight of hazardous emissions per unit weight of product.

Source: Navinchandra, 1991.

nondegradable portions may be to the environment. The approach merely assumes that less nondegradable material is necessarily "better" for the environment.

Some additional weaknesses of the green indicators include the following:

- too simplistic,
- does not account for impacts to human health, and
- unclear how some indicators (e.g., life-cycle cost) would be calculated.

Relevance to Impact Assessment

The green indicators approach would likely be most suitable for a less detailed Tier 1-type assessment of environmental impacts. Although somewhat simplistic, the green indicators would enable decisionmakers to consider a wide variety of factors in addition to emission levels that can be integral to making decisions concerning tradeoffs between alternative products, processes, and materials as well as between environmental, economic, and production concerns.

B.2 POLAROID'S ENVIRONMENTAL ACCOUNTING AND REPORTING SYSTEM (EARS)

Polaroid's Environmental Accounting and Reporting System (EARS) was developed as a tool to help measure the progress of its Toxic Use and Waste Reduction (TUWR) Program goals. EARS is a centralized database that allows Polaroid to track virtually every one of the 1,400 materials the company uses, from office paper to chlorinated solvents (Nash et al., 1992). Each material is classified into one of five toxicity categories to reflect the degree of potential environmental harm it poses (see Table B-2). With EARS, Polaroid records the quantities and treatment methods of materials in all five categories at several points along the process line. Use, waste, and by-products are measured and recorded per unit of production.

Strengths

EARS has turned out to be a beneficial program because it

- provides employees with information needed to assess the environmental quality of their actions;
- provides incentives for making continual improvements in environmental performance;
- provides an effective Total Quality Environmental Management (TQEM) tool, fulfilling several different functions throughout the company;
- allows employees to predict the environmental impacts of new chemicals before the company makes a commitment to their use; and
- translates complex environmental data into a simple index that has meaning throughout the company (Nash et al., 1992).

TABLE B-2. POLAROID'S EARS CATEGORIZATION OF CHEMICALS

Category	Number of Chemicals	Examples	Environmental Impact	Reduction Emphasis
I & II	Category I – 38 Category II – 65	• ammonia • benzene • CFCs	Most severe environmental impact; highly toxic; human carcinogens	Minimize use
III		• acetic acid • pyridine • styrene	Moderately toxic; corrosive; suspected animal carcinogens	Recover and reuse onsite
IV	All remaining chemicals	• acetone • butanol	Least environmental impact	Reuse onsite following on or offsite recycling
V		• cardboard • paper • plastic	Depletes natural resources during manufacture and disposal	Maximize recycling and reuse onsite

Source: Modified from Nash et al., 1992.

Weaknesses

The primary weakness of EARS is that it does not measure environmental releases nor does it estimate environmental impacts. EARS is essentially a classification system in which chemicals may be grouped according to their known environmental toxicity.

In addition, many complain that EARS data requirements are too time consuming and that EARS is cumbersome to use (see Nash et al., 1992). Accuracy is also a persistent concern. People responsible for computing EARS numbers and recording the data have varying levels of skill and familiarity with the materials of interest. In addition, EARS is not linked with the company's financial system. Thus, the company is unable to readily assess the financial benefits of environmental improvements to its operation.

Relevance to Impact Assessment

It is unclear how EARS could be used in the context of impact assessment. Perhaps at a most basic level, inventory items could be grouped into EARS-like categories based on their relative environmental toxicity. This would result in a listing of the most critical inventory items

and their respective quantities that possibly could help analysts and decisionmakers pinpoint improvement opportunities and/or areas that require a more detailed level of analysis. Such an approach would likely be more appropriate for internal rather than external applications.

B.3 JUDGMENT PROBABILITY ENCODING

Judgment probability encoding was developed by Argonne National Laboratory to provide a means of quantifying subjective probabilities for impacts. The main objective of this approach is to reduce divergence among expert judgment through an encoding process in estimating the probability of impact(s) resulting from exposure to substances.

Encoding in this context ensures that the questions used to derive judgment probabilities are always phrased identically, that specific assumptions and definitions are always the same, and that the encoding process proceeds similarly for each of the participants (Argonne National Laboratory, 1991). Thus, any differences in judgment probabilities can be attributed to true differences in values or opinions and not to differences in assumptions, understanding, or procedures.

The output of the judgment probability encoding approach is a range of probabilities regarding a specific function, (i.e., the likelihood of impact X resulting from pollutant A). The encoded judgment probability may then be communicated in a variety of ways—as a distribution, a range, a mean, or a median. For example, consider the scenario where five experts were solicited for judgment probabilities regarding the likelihood of X tons of CFCs being linked with stratospheric ozone depletion. Each expert is provided exactly the same information, assumptions, understanding, and procedures in exactly the same manner. For the purposes of this example, generic probabilities are provided in Table B-3. These judgment probabilities are then used to derive further quantitative characterizations of value judgments. For instance, the analyst may decide to use a mean (0.25) to express the probability judgment values or a range (0.15 to 0.35).

Strengths

The primary advantage of the judgment probability encoding approach is that it can take into account the normalization (via impact probability values) of a wide variety of potential impacts. By normalizing impacts, this approach enables decisionmakers to choose alternatives from a subjective point of view—by relying only on the probability figures as impact descriptors. In addition, the judgment probability encoding approach is easy to conduct.

TABLE B-3. GENERIC ENCODED JUDGMENT PROBABILITIES EXERCISE

Expert	Judgment Probabilities for the Occurrence of Impact A
1	0.20 – 0.25
2	0.10 – 0.15
3	0.20 – 0.30
4	0.15 – 0.20
5	0.25 – 0.35

Median = 0.20 – 0.25
Mean = 0.18 – 0.25
Range = 0.10 – 0.35

Weaknesses

The disadvantages of the judgment probability encoding approach are that it measures impacts indirectly, in terms of judgment probabilities, and it may be too simplistic for impact assessment. It would also, for all practical terms, be impossible to replicate the results of a judgment probability encoding study. However, results from similar studies could be used to verify and support the results of a judgment probability encoding study.

A code of good practice will need to be established for selecting and conducting the expert encoding process to elicit judgment probabilities. Some questions that may need to be considered in this respect include the following:

- Who chooses the expert panel?
- How many experts are required to conduct the approach?
- From which fields should the experts be chosen?
- Who approves the selection of experts and monitors the judgment probability encoding process?

Relevance to Impact Assessment

The judgment probability encoding process may be useful in the context of impact assessment as a simplified impact characterization approach based upon expert judgment. Being an entirely subjective approach, it would be more appropriate for internal that external

applications. In addition, the judgment probability encoding approach may be useful in cases where data on environmental conditions are not available or where nontraditional impact categories are involved (e.g., species loss, habitat destruction, aesthetic loss).

B.4 HUMAN EXPOSURE DOSE/RODENT POTENCY DOSE INDEX

The Human Exposure Dose/Rodent Potency Dose (HERP) Index provides a common factor for measuring the potency of various carcinogenic substances. The HERP Index is calculated by determining the ratio of TD_{50} to human exposure. TD_{50} is the daily dose rate (in milligrams per kilogram) needed to halve the percentage of tumor-free animals at the end of a standard lifetime (Ames et al., 1987). Analogous to $LD_{50,}$ the lower the dose rate, or TD_{50} value, the more potent the carcinogen. Some example HERP Index values for specific carcinogens are shown in Table B-4. Because the rodent data are calculated on the basis of lifetime exposure at the indicated daily dose rate, the human exposure data are also expressed as lifetime daily dose rates despite the notion that human exposure may likely be less than daily over a lifetime.

TABLE B-4. EXAMPLE HERP INDEX VALUES

HERP (%)	Daily Human Exposure	Carcinogen Dose Per 70-kg Person	Potency of Carcinogen: Rats	Mice	References
0.001	1 liter (tap water)	Chloroform, 83 μg	(119)	90	96
0.004	1 liter (well water—worst)	Trichloroethylene, 2,800 μg	(-)	941	97
0.0004	1 liter (well water—best)	Trichloroethylene, 267 μg	(-)	941	948
0.0002		Chloroform, 12 μg	(119)	90	
0.0003		Tetrechloroethylene, 21 μg	101	(126)	
0.008	1 hour (pool)	Chloroform, 250 μg	(119)	90	99
0.6	14 hours (A/C conventional home)	Formaldehyde, 598 μg	1.5	(44)	100
0.004		Benzene, 155 μg	(157)	53	
2.1	14 hours (A/C mobile home)	Formaldehyde, 2.2 μg	1.5	(44)	28

Source: Ames et al., 1987.

Strengths

Using the HERP Index to assess carcinogenic impacts provides a means of normalizing carcinogenic substances and allows for different types of carcinogens to be directly compared for their carcinogenic potential. In addition, the HERP Index allows for different types of carcinogens to be aggregated so that the total contribution of inventory items to cancer can be assessed. In addition, a TD_{50} database already exists but is quite extensive.

Weaknesses

On the downside, using the HERP Index values as direct estimates of impact would be inappropriate. Many uncertainties and assumptions are associated with extrapolating from experimentation on rodents to values for human carcinogenicity. Another problem with using the HERP Index is that information is lacking on natural carcinogens and their relationship to man-made carcinogenic substances.

In addition, the HERP Index is based on the assumption that dose-response relationships are linear, but this assumption may not be correct. Dose responses that are not linear but quadratic or hyperbolic would yield HERP Index values much lower than those obtained by using a linear dose response mechanism.

Relevance to Impact Assessment

The HERP Index may be useful in the context of impact assessment for characterizing, comparing, and/or aggregating the carcinogenic impact of inventory items. It should be stressed that this method is only applicable for assessing carcinogenic impacts. However, because the HERP index is highly controversial within its own field of human health research, it should not be used in impact assessment.

B.5 ENVIRONMENTAL INDICES

A wide variety of environmental indices have been developed to provide an estimate ambient pollutant levels in different environmental media. These ambient levels of pollutants are used as a proxy for estimating environmental impacts. These indices are, in essence, equivalency functions that may be used to compare the relative impact of a variety of different substances released into the environment. This section discusses two main groups of indices—air pollution indices and water pollution indices.

Strengths

A main strength of the environmental indices described in this section is that they have been developed, refined, and used in practice for a number of years. There is a large body of experience to draw upon for using and interpreting such indices.

Weaknesses

A primary weakness of the indices included in this section is that they account for only a small subset of possible pollutants. In addition, the indices provide measures of ambient concentrations for a region as a whole. Thus using the indices to estimate the contribution of a single source of pollution to overall regional levels would be difficult.

Relevance to Impact Assessment

Although the indices described in this section do not measure impacts per se, they may be used to compare "before" and "after" scenarios for the releases of a proposed project or used as baseline information for conducting a detail impact assessment. Beyond providing an indication of ambient pollutant concentrations in regional air and water sinks, the use of environmental indices in the context of impact assessment is unclear.

B.4.1 Air Pollution Indices

A number of air pollution indices have been proposed in journals, conference proceedings, and research reports. Additional indices have been developed by state and local air pollution control agencies and have been implemented to routinely report air quality data to the public. In the mid-1970's, so many different reporting schemes were in use that the government found it necessary to adopt a national air pollution index, the Pollutant Standards Index (PSI) (Ott, 1987). The PSI and other air pollution indices are summarized in Table B-5.

B.4.2 Water Pollution Indices

Indices have also been developed that can be used with data available from current water quality monitoring activities to provide an estimate of water pollution (see Table B-6). There are two basic types of water pollution indices: increasing scale indices and decreasing scale indices. Increasing scale indices refer to "water pollution" indices while decreasing scale indices refer to "water quality" indices (Ott, 1987). Water pollution indices may also be grouped into five main categories:

TABLE B-5. CLASSIFICATION OF AIR POLLUTION INDICES

		Variables[b]						
Index	Classification[a]	CO	NO_2	OX	TSP	COH	SO_2	Other
Green's Index	$2A_3C$					•	•	
Combustion Products Index (CPI)	$2C_1C$							c
Measure of Undesirable Respirable Contaminates (MURC)	$1A_1C$					•		
Air Quality Index (AQI)	$3C_3C$	•			•		•	
Ontario Air Pollution Index (API)	$2A_3B$					•	•	
PINDEX	$7C_3C$	•	•	•	•		•	d
Oak Ridge Air Quality Index (ORAQI)	nA_3A (n=1 to 5)	•	•	•	•		•	
MITRE Air Quality Index (MAQI)	nA_3A (n=1 to 5)	•	•	•	•		•	
Extreme Value Index (EVI)	nA_3A (n=1 to 4)	•		•	•		•	
Short Time Averaging Relationships to Air Quality Standards (STARAQS)	$6B_3A$	•	•	•	•	•	•	
Environmental Quality Index (EQI-air)	$8A_3A$	•	•	•	•	•	•	e
Pollution Standards Index (PSI)	$5B_2B$	•	•	•	•		•	
	Total	7	5	6	7	5	9	

[a] Classification is based on the Thom-Ott air pollution index classification system. The first digit indicates the number of pollutants the index addresses. The first letter indicates the calculation method used, where A = nonlinear, B = segmented linear, C = linear, and D = actual concentrations. The subindex number to the calculation method indicates the type of calculation model used, where 1 = individual, 2 = maximum, and 3 = combined. The last letter indicates the type of descriptor categories used by the index, where A = standards, B = standards and episode criteria, and C = arbitrary.

[b] CO, carbon monoxide; NO_2, nitrogen dioxide; OX, photochemical oxidants; COH, coefficient of haze; SP, total suspended particulates; SO_2, sulfur dioxide.

[c] Fuel burned and ventilating volume.

[d] Hydrocarbons and solar energy.

[e] Visibility and industrial emissions.

Source: Ott, 1987.

TABLE B-6. CLASSIFICATION OF WATER POLLUTION INDICES

Index	Number of Variables	Scale	Variables Used
General Indices			
Quality Index (QI)	10	decreasing	DO, alkalinity, chlorides, CCE, pH, temperature, specific condition, total coliforms, other biological.
Water Quality Index (WQI)	9	decreasing	DO, BOD, nitrates, phosphates, pH, temperature, turbidity, total solids, fecal coliforms.
Implicit Index of Pollution	13	increasing	DO, BOD, COD, iron, manganese, ammonia, nitrates phosphates, ABS, CCE other chemical, pH, suspended solids.
River Pollution Index (RPI)	8	increasing	DO, BOD, COD, phosphates, other chemical, temperature, specific condition, total coliforms.
Social Accounting System	11	decreasing	DO, BOD, alkalinity, hardness, chlorides, pH, temperature, specific condition, total solids, fecal coliforms, total coliforms.
Specific-Use Indices			
Fish and Wildlife (FAWL) Index	9	decreasing	DO, ammonia, nitrates, phosphates, phenol, pH, temperature, turbidity, dissolved solids.
Public Water Supply (PWS) Index	13	decreasing	DO, alkalinity, hardness, nitrates, chlorides, fluorides, sulfates, phenol, pH, turbidity, dissolved solids, color, fecal coliforms.
Index for Public Water Supply	11/13	decreasing	DO, BOD, hardness, iron, nitrates, fluorides, phenol, pH, temperature, turbidity, dissolved solids, color, fecal coliforms.
Index for Recreation	12	decreasing	DO, nitrates, phosphates, oil and grease, pH, temperature, turbidity, suspended solids, color, other physical, total coliforms.
Index for Dual Water Uses	31	decreasing	iron, manganese, ammonia, nitrites, chlorides, fluorides, sulfates, phenol, other chemical, pH, specific condition, color, fecal coliforms.

(continued)

TABLE B-6. CLASSIFICATION OF WATER POLLUTION INDICES (CONTINUED)

Index	Number of Variables	Scale	Variables Used
Specific-Use Indices (continued)			
Index for Three Water Uses	14	increasing	DO, alkalinity, hardness, iron, manganese, chlorides, sulfates, pH, temperature, turbidity, suspended solids, total solids, color, other physical, fecal coliforms.
Planning Indices			
Prevalence Duration Intensity (PDI) Index	b	increasing	*Note: Because of their flexibility and special-purpose nature, the planning indices and statistical approaches do not lend themselves to detailed comparison.*
Nation al Planning Priorities Index (NPPI)	b	increasing	
Priority Action Index (PAI)	b	increasing	
Environmental Evaluation Systems (EES)	78[a]	decreasing	
Canadian Pollution Index (CPI)	b	increasing	
Potential Pollution Index (PPI)	3	increasing	
Pollution Index (PI)	b	increasing	
Statistical Approaches			
Composite Pollution Index (CPI)	18	increasing	
Index of Partial Nutrients	5	decreasing	
Index of Total Nutrients	5	decreasing	
Principal Component Analysis	b	N/A	
Harkins' Index	b	increasing	
Beta Function Index	b	increasing	

[a] Water quality variables account for 14 of the 78 variables used in this system.
[b] Any number of variables can be included.

Source: Ott, 1987.

- general water quality indices,
- specific-use indices,
- planning indices,
- statistical approaches, and
- biological indices (Ott, 1987).

Table B-6 summarizes these indices (with the exception of biological indices not amenable to classification). Three general types of biological water quality indices evaluate water quality on the basis of its impact on aquatic life—types and quantities of certain indicator organisms, mathematical properties of populations of organisms, and physiological or behavioral responses of certain organisms to pollution.

B.6 DEGREE OF HAZARD EVALUATION

The degree of hazard evaluation system was developed as a scientifically sound and consistent way to deregulate the tracking of non-RCRA special wastes that pose low or negligible hazard. The degree of hazard evaluation ranks wastes according to their respective degrees of hazard and is based on five characteristics of a waste stream:

- weighted accumulative toxicity of constituents (as modified by environmental fate),
- disease potential (infectious waste),
- fire (ignitability),
- leaching agents (pH), and
- biological hazard (biodegradability) (Plewa et al., 1986).

The degree of hazard evaluation places primary emphasis on toxicity to rank potential hazard. Thus toxicology data are used to generate a numerical score for a substance's equivalent toxicity (Plewa et al., 1986). The calculation of equivalent toxic concentration of each life-cycle waste component (C_{eq}) is as follows:

$$\text{Equivalent Toxic Concentration} = C_{eq} = A \sum_i (C_i / B_i T_i)$$

where

C_i = the concentration of component i as a percentage of the waste by weight,

T_i = a measure of the toxicity of component i,

A = a constant equal to 300 used to allow entry of percent values for C_i and to adjust the results so that a reference material, 100 percent copper sulfate with an oral toxicity of 300 mg/kg, achieves an equivalent toxicity of 100, and

B_i = a conversion factor used to convert toxicities (t_i) to equivalent oral toxicities.

Table B-7 shows conversion factors (B_i) for various toxicity measures.

For carcinogens and mutagens, a TD_{50} oral rat dose is used if available. Otherwise carcinogens are assigned a T_i 0.1 mg/kg, and mutagens are assigned a T_i of 0.6 mg/kg. Toxicities are converted to equivalent oral toxicities as specified in Table B-7. Oral rat toxicity values are preferred, followed by inhalation rat, dermal rabbit, aquatic toxicity, and other mammalian toxicity values. If there is more than one value for the toxicity from the best available source, the lowest (most toxic equivalent oral toxicity value) is used. If a carcinogen or mutagen is assigned a value for T_i in the absence of a TD_{50}, B_i is assigned a value of 1.

The relative toxic amount, M, of the entire waste stream mixture is calculated as follows:

$$\text{Relative Toxic Amount} = M = S\ C_{EQ}$$

where S = the maximum size (kg) of waste output produced in a month. The result of these calculations will be an estimate of the relative toxic amount (M) for each waste output evaluated that takes into account the comparative toxicity and amount of each component. For each waste output, the number calculated for M can range from 0 to greater than 10,000. The relative toxic amount is then converted into categories of hazard: negligible, low, moderate, or high.

TABLE B-7. TOXICITY CONVERSION FACTORS

Conversion Factors For The Equivalent Oral Toxicities (B_i):		
Toxicity Measure	**Units**	**B_i**
Oral – LD_{50}	mg/kg	1.00
Carcinogen/mutagen – LD_{50}	mg/kg	1.00
Aquatic – 48 or 96 hr LC_{50}	ppm	5.00
Inhalation – LC_{50}	mg/l	25.00
Dermal – LD_{50}	mg/kg	0.25

Source: Thomas and Miller, 1992.

The results of an actual degree of hazard evaluation conducted to evaluate two types of sand wastes produced by an iron foundry are illustrated in Tables B-8 and B-9. For a more complete description of the degree of hazard evaluation, refer to Reddy (1985), Plewa et al. (1986), and Plewa et al. (1988).

Strengths

The degree of hazard evaluation method may be used to normalize chemical substances in a manner that allows the analyst to compare not only the equivalent toxicity of various chemicals but also other inherent characteristics of those hemicals. In addition, the degree of hazard evaluation method has been used in practice and refined for a number of years.

Weaknesses

One problem with using the degree of hazard evaluation in impact assessment is data availability. Out of over 5,000 RCRA and non-RCRA waste streams analyzed, over 70 percent were ranked as "unknown" hazards due primarily to the following data deficiencies:

- information that was required on waste streams but was missing,

TABLE B-8. DEGREE OF HAZARD EVALUATION OF IRON FOUNDRY MOLDING SAND WASTE #1

	Sand Waste #1	
Component Name	**Concentration (5)**	**Equivalent Toxicity**
Chromium	0.000002	0.00006
Barium peroxide	0.000012	0.000003
Arsenic pentoxide	0.000002	0.00000008
Lead monoxind	0.000005	0.000000001
Cadm ium	0.000000	0.000000000
Selenium dioxide	0.000002	0.000000000
Total Equivalent Toxicity		**0.000063**
Overall Hazard Ranking		**Negligible**

Source: Thomas and Miller, 1992.

TABLE B-9. DEGREE OF HAZARD EVALUATION OF IRON FOUNDRY MOLDING SAND WASTE #2

	Sand Waste #2	
Component Name	**Concentration (%)**	**Equivalent Toxicity**
Nickel	0.00171	0.0513
Phenol	0.001544	0.00772
Cadmium	0.00008	0.0024
Chl oroform	0.000039	0.00117
Barium peroxide	0.00028	0.000067
Fluorine	0.09	0 .000058
Chromium oxide	0.00017	0.000006
Lead monoxide	0.000074	0.000000964 3
Xylenes, total	0.000002	0.0000000888
Arsenic pentoxide	0.000002	0.000 000075
Methylene chloride	0.00003	0.00000005389
Toluene	0.000044	0.0000 000528
2-butanone	0.00022	0.00000004074
Acetone	0.00042	0.0000000252
Silver dioxide	0.000035	0.00000000372
Mercury oxide	0.000000000	0.00000 0000
Selenium dioxide	0.000003	0.000000000
Silica	99.9049	0.000000000
Total Equivalent Toxicity		**0.0068**
Overall Hazard Ranking		**High**

Source: Thomas and Miller.

- data necessary for many toxicity hazard calculations that were not available in the public literature, and
- vague names for wastes or chemicals that were often used rather than trade names.

Relevance to Impact Assessment

The degree of hazard evaluation method may be used to normalize a variety of chemical-based inventory items in a manner that allows the analyst to compare not only the equivalent

toxicity of various inventory items but also other inherent characteristics of those items as well. In addition, degree of hazard evaluation projects have been used in practice for a number of years and thus may be currently applicable to impact assessment.

B.7 HAZARD RANKING METHODS

A number of hazard ranking methods have been developed for a variety of different purposes. Hazard ranking methods, much like Tier 2- and Tier 3-type characterization models, rank the relative risk of substances released to the environment based on hazard (e.g., toxicity) and sometimes exposure (e.g., persistence, bioaccumulation) information. The following sections describe some of the primary hazard ranking methods.

Strengths

Hazard ranking methods have several identifiable advantages. Most are relatively easy to use, they do not require extensive data, and three major routes (groundwater, surface water, and air) are considered. In addition, factors have been carefully selected for consistency and to avoid redundancy, and they often are built upon previously developed models (including the JRB Associates, Inc. model).

Weaknesses

A number a criticisms have been raised about using hazard ranking methods:

- The score for hazard potential is based on only the most hazardous substance rather than on a composite of all constituents.
- Low population areas tend to receive lower scores than higher population areas.
- The use of distance to population as a weighting factor is used even in situations where there is no evidence of release.
- Few provisions exist for incorporating additional technical information into the models.
- Individual factor scores are often aggregated into a composite total score.

Relevance to Impact Assessment

The hazard ranking methods described in this section are most similar to the Tier 2- and Tier 3-type characterization models described in Chapters 3 and 4 (see the toxicity, persistence, and bioaccumulation assessment). Most of these methods have scoring systems in which the relative "hazard" associated with a variety of substances is estimated. Because most of the hazard ranking methods focus on impacts to human health, it is not clear whether they would be useful for estimating impacts to ecosystems or natural resources.

B.7.1 EPA's Hazard Ranking System (HRS)

The HRS was developed by the MITRE Corporation to meet Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) requirements mandating that ranking systems be based on relative risks. In this context, relative risk takes into account the population at risk, the hazardous potential of releases, the potential for contamination of drinking water supplies (for both ecosystem and human health impacts), and other appropriate factors. HRS ranks facilities in terms of the potential threat they pose by describing the manner in which hazardous wastes are contained, the route by which they are released, the characteristics and amount of the hazardous substance, and the likely ecosystem and human health targets (see Table B-10).

TABLE B-10. OVERVIEW OF RATING FACTORS

	Factors		
Category	**Groundwater Route**	**Surface-Water Route**	**Air Route**
Route Characteristics	• depth to aquifer of concern • net precipitation • permeability of unsaturated zone • physical state	• facility slope and intervening terrain • one-year 24-hour rainfall • distance to nearest surface water • physical state	
Containment	• containment	• containment	
Waste Characteristics	• toxicity/persistence hazardous waste quantity	• toxicity/persistence hazardous waste quantity	• reactivity • incompatibility • toxicity • hazardous waste quantity
Targets	• groundwater use • distance to nearest well/population served	• surface water use • distance to sensitive environment • population served/distance to water intake downstream	• land use • population within 4-mile radius • distance to sensitive environment

Source: Federal Register, 1988.

The HRS assigns three scores to a hazardous facility:

1. The potential for harm to humans or the environment from migration of a hazardous substance away from the facility by routes involving groundwater, surface water, or air.
2. The potential for harm from substances that can explode or cause fires.
3. The potential for harm from direct contact with hazardous substances at the facility (Sandia National Laboratories, 1986).

Scores for each hazard mode are determined by evaluating a set of factors that characterize the potential of the particular facility to cause ecosystem and human health impacts. Each factor is assigned a numerical value on a scale of 0 to 3, 5, or 8, according to prescribed guidelines. The assigned value is then multiplied by a weighting factor to yield the individual factor score. The individual scores may then be aggregated within each factor category, and then the aggregated scores for each factor category are multiplied together to develop scores for migration (groundwater, surface water, air), fire and explosion, and direct contact.

Use of the HRS requires information about the facility in question, its surroundings, the hazardous substances present, and the geological characteristics of the surrounding area. When there are no data for a factor, it is assigned a value of zero. However, if a factor with no data is the only factor in a category, then the factor is given a score of 1.

B.7.2 Modified Hazard Ranking System (MHRS)

The Modified Hazard Ranking System (MHRS) was developed by Battelle Pacific Northwest Laboratory (PNL) for DOE to rank sites that contain both chemically hazardous and radioactive wastes. MHRS was developed to work within the framework of EPA's HRS, and the overall scoring system is the same for both methods. The modifications to the HRS for sites containing radioactive wastes were restricted to the waste characteristics category of the groundwater, surface-water, air, fire and explosion, and direct-contact routes.

In developing a scoring system for radioactive wastes in MHRS, the concentration and the type of radiation emitted by the radionuclides were factored into the ranking. The scoring of the radionuclides is based on an estimate of the potential radiation dose to a maximally exposed individual (the product of dose factor times concentration is estimated).

The MHRS splits the waste characteristics categories into chemical wastes and radioactive wastes. The scoring system for chemical wastes is the same as that of EPA's HRS. The hazards of the radioactive and nonradioactive wastes are evaluated separately and the score

is assigned over the same range of values. The higher score of the two is the value assigned to the site. The site ranking is based on the maximum score (chemical or radioactive) from each route and is calculated as described in HRS. Scoring for radioactive wastes through each route is described below.

For the air route, information on the maximum observed concentration of radionuclides in air at the site is required. If no concentration of atmospheric radioactivity significantly above background has been observed, then the waste characteristics score for the air route is zero. If release of radionuclides has been observed, then the total concentration for each radionuclide group is calculated. A matrix table for the air route is then used to determine the waste characteristics score by selecting the largest value among the groups.

For the surface-water route, if release has been observed, the total surface-water concentration for each nuclide group is determined and the highest resulting waste characteristics score among the groups is selected. The largest score among nuclide groups derived from the maximum potential surface-water releases is then compared with that from observed release. The greater of the two is recorded in the surface-water route.

In the groundwater route, if release has been observed, the highest waste characteristics score among the nuclide groups resulting from the observed releases is used. This score is then compared with the score calculated from the maximum potential release. The maximum potential concentration for each radionuclide is determined by multiplying the amount disposed of at the site by the transport coefficient. The total potential groundwater concentration associated with each nuclide group is calculated by summing all radionuclides within the group (see Table B-11). The waste characteristics score for each nuclide group can then be determined from a matrix table (see Table B-12). The largest value among the groups is compared with that from the observed release. The greater of the two values is recorded for the groundwater route.

The fire and explosion route and the direct-contact route are usually of less importance than other routes for hazardous waste sites. Therefore, a detailed description for scoring these two routes is not provided here.

B.7.3 U.S. Air Force (USAF) Hazard Assessment Rating Methodology

The USAF has sought to establish a system to "develop and maintain a priority listing of contaminated installations and facilities for remedial action based on potential hazard to public health, welfare, and environmental impacts" (Sandia National Laboratories, 1986). As part of this system priorities are to be set for taking further actions at sites. Thus the Hazard

TABLE B-11. RADIONUCLIDE GROUPS

Group	Nuclides
A	$^{226+D}Ra$, unidentified alpha emitters
B	^{129}I, $^{210+D}Pb$, ^{90+D}Sr, ^{229}Th, $^{233+D}U$, unidentified beta and gamma emitters
C	^{241}Am, ^{243}Am, ^{134}Cs, $^{237+D}Np$, $^{230+D}Th$, $^{232+D}Th$
D	^{243}Cm, ^{244}Cm, ^{60}Co, ^{135}Cs, ^{17+D}Cs, ^{152}Eu, ^{154}Eu, ^{22}Na, ^{94}Nb, ^{73}Ni, ^{238}Pu, ^{239}Pu, ^{240}Pu, ^{234}U, ^{244}Pu , ^{225}Ra, ^{151}Sm, ^{99}Tc, $^{228+D}Th$, ^{234}U, $^{238+D}U$, $^{235+D}U$
E	^{225}Ac, ^{14}C, ^{55}Fe, ^{93}Mo, ^{59}Ni, ^{239}Np, ^{241}Pu, $^{125+D}Sb$, ^{240}U
F	^{3}H

Source: Sandia National Laboratories, 1986

TABLE B-12.MATRIX TABLE FOR GROUNDWATER ROUTE WASTE CHARACTERISTICS SCORE

	Maximum Ground-Water Concentration (pCi / L)												
Nuclide	10^{-3}	10^{-2}	10^{-1}	10^{0}	10^{1}	10^{2}	10^{3}	10^{4}	10^{5}	10^{6}	10^{7}	10^{8}	10^{9}
A	0	1	3	7	11	15	21	26					
B		0	1	3	7	11	15	21	26				
C			0	1	3	7	11	15	21	26			
D				0	1	3	7	11	15	21	26		
E					0	1	3	7	11	15	21	26	
F						0	1	3	7	11	15	21	26

Source: Sandia National Laboratories, 1986

Assessment Rating Methodology was developed to provide a relative ranking of sites that are suspected of having been contaminated from hazardous substances.

The Hazard Assessment Rating Methodology considers four aspects of the hazard posed at a specific site:

- the possible receptors of the contamination,
- the waste and its characteristics,
- potential pathways for waste contaminant migration, and
- any efforts to contain the contaminants.

Each of these categories contains a number of rating factors that are used in the overall hazard rating. For example, the waste characteristics category is scored in three steps. First, a point rating is assigned based on an assessment of the waste quantity and the hazard (worst case) associated with the site. Second, the score is multiplied by a waste persistence factor, which reduces the score if the waste is not very persistent. Finally, the score is modified according to the physical state of the waste. Scores for liquid wastes are unchanged, while scores for sludges and solids are reduced. The scores for each of the three categories are then added and normalized to a maximum possible score of 100.

The USAF Hazard Assessment Rating Methodology is based on the same JRB model as the EPA HRS method and is similar in many respects. The best way to highlight the strengths and weaknesses of the USAF ranking method is to identify those components that differ significantly from the HRS approach. These differences are found in the areas of:

- *Waste Quantity*: the USAF method deals more realistically with the quantities of toxic substances by having "quantity" indicate the total amount of chemicals in a particular hazard classification.
- *Persistence*: values for persistence in the USAF method are used to modify the waste characteristics score (based on toxicity and quantity). This may be inappropriate because different types of chemicals contribute to the overall waste characteristics score (i.e., it is better to combine toxicity and persistence considerations for individual chemicals as done in the HRS than to apply one persistence score to a diverse class of chemicals).
- *Air Releases*: these are not considered in the USAF ranking method, thus the potential risks associated with a site could be underestimated.

B.7.4 Relative Hazard Ranking System

Hazard evaluations for toxic chemicals and low-level radioactive wastes have generally been performed independently of one another and without a means of comparison. Exposure of ecological systems to ionizing radiation usually results in nonspecific damage, while exposure to a chemical can produce specific damage to a specific biologic activity. Comparing radioactive hazards with chemical hazards is difficult because of differences between the underlying

mechanisms of radiation and chemical effects. In ranking waste disposal sites that contain a mix of chemical and radioactive wastes, a relative rating of chemical hazard and radiation effects is necessary. A few approaches have been suggested that might be useful in comparing relative hazards. Four of these approaches are summarized in Table B-13.

B.8 THE AMOEBA APPROACH

AMOEBA is the Dutch acronym for "a general method of ecosystem description and assessment." The AMOEBA approach is based on the concept of sustainable development and was developed for and applied to the Dutch Water Management Plan (see Kuik and Verbruggen, 1991; and Udo de Haes, Nip, and Klijn, 1991). AMOEBA is a conceptual model for the development of quantitative and verifiable ecological objectives, and it provides a means for quantitatively describing and assessing ecosystems.

The AMOEBA approach employs "ecological values," which are defined as desired states of ecological components as predetermined by decisionmakers and/or stakeholders. In order to establish precisely these ecological values, the most fundamental values humans attribute to plant and animal life are examined. Three categories of ecological characteristics are used in deriving ecological values:

TABLE B-13. ALTERNATIVE APPROACHES TO RELATIVE HAZARD RATINGS

Approach	Applicability	Limitations
Rem-Equivalent Chemical	• carcinogenic • mutagenic • teratogenic • substances	The significance of dose-response and safety standards is undefined and depends on levels of acceptable risk.
MPC/EPC-Air and Water Equivalents	• performance criteria • disposal volumes • offsite concentration limits	Depends on validity of MPC/EPC limits subject to change.
Equivalent Hazard Categories	• general toxic effects • based on definitive data	Database is usually acute rather than chronic toxicity.
Site-Specific Risk Management Committee	• local conditions • credible • easily understood	Potentially subjective changes in value judgments with time or committee members.

- **Production and Yield:** These characteristics are valuable for functional reasons. This category is a prerequisite for human existence (e.g., fisheries). These values are closely associated with the abundance of species, the production of oxygen, and the self-purifying capacity.
- **Species Diversity:** This is valuable for ethical and aesthetic considerations. It involves concepts such as the preservation of species, rarity, and completeness.
- **Self-Regulation:** Self-regulation has ethical, aesthetic/recreational, and economic considerations that are closely related to concepts such as naturalness, stability, intactness, authenticity, and visual integrity. Moreover, self-regulating ecosystems have low management costs (Kuik and Verbruggen, 1991).

AMOEBA-type approaches typically present three values for each study parameter: reference (baseline) values, target (objective) values, and current (measured) values. The relationship between these three values is shown in Figure B-1. These values are plotted on a circular figure for each parameter (see Udo de Haes, Nip, and Klijn, 1991). Determination of these three values is integral to the AMOEBA approach.

Reference values are obtained by a reference system, which has been only slightly influenced by human activities or not at all. Such a system contains the conditions for the evolution and survival of organisms, including humans, living in and around it. The introduction of a reference system provides a standard against which an assessment of the ecological condition of a system can be made. The closer one can come to mimicking the reference system, the larger the chance of ecological sustainability. The overall ecological objective, however, does not necessarily have to coincide with the reference system.

Decisionmakers and/or stakeholders must decide on the maximum acceptable distance from the reference point to establish a verifiable ecological objective. This distance is the target value. Target values may both exceed or fall short of the reference values, depending on the parameter. The compromise between the ecological quality objective (the target value) and the reference value is evidenced by the discrepancies between the two values.

Current or measured values represent the actual state of the system. Current values may be determined by direct measurement, modeling, or through secondary data sources. The difference between the target values and the current value indicates the extent to which the ecological objective has fallen short of or been surpassed by either an existing or proposed activity.

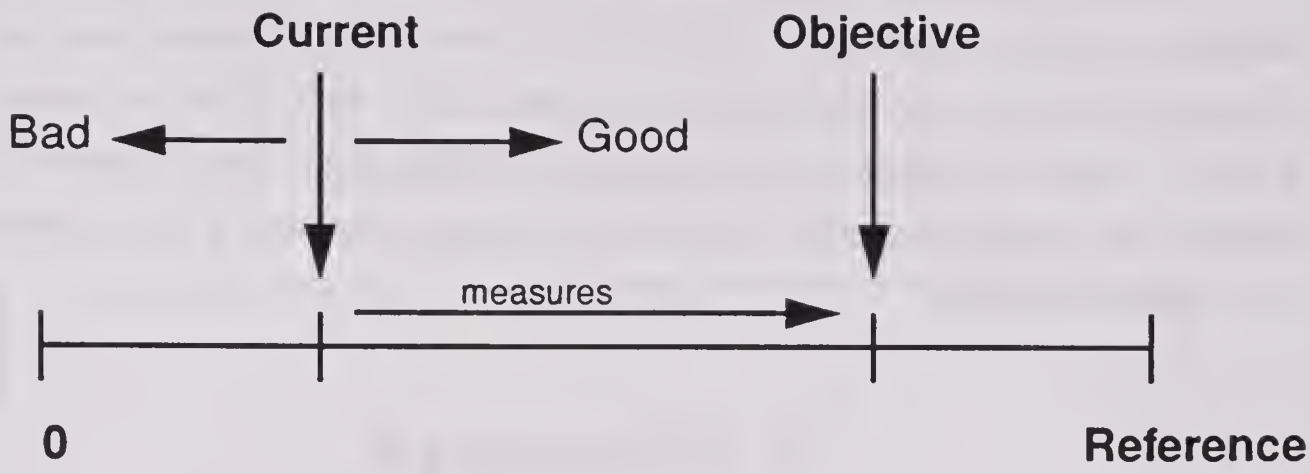

Figure B-1. Relationship Between AMOEBA Components

Source: Udo de Haes, Nip, and Klijn, 1991.

A case study example using the AMOEBA approach is provided in Udo de Haes, Nip, and Klijn (1991).

Strengths

The primary strength of the AMOEBA approach is that it uses actual environmental conditions as baseline information against which to estimate the environmental implications of a project. Such an assessment can provide a more realistic study of the effects of a project on the environment.

Weaknesses

A weakness of the AMOEBA approach is that, although reference environmental values are determined relatively objectively, establishing target values is a highly subjective process that would involve consulting experts from a variety of different fields of expertise. In addition, a method for integrating and interpreting the effects of environmental releases from other projects is not clear.

Relevance to Impact Assessment

In the context of impact assessment, the AMOEBA approach may be useful for estimating the extent to which impacts are within or exceed the stakeholder-determined maximum acceptable values (the target values) from environmental reference points.

In addition, because the current values represent the actual state of the system, the difference between the target values and the current value indicates the extent to which the ecological objective has fallen short of or been surpassed by either an existing or proposed activity. This information may be useful for highlighting to decisionmakers the compromise between the ecological quality objective (the target value) and ambient environmental conditions (the reference value).

Appendix C

Key Terms and Definitions

Assessment Endpoint	An impact of concern identified from a variety of potential impacts resulting from any given inventory item. Assessment endpoints are determined based on the goals and scope of the LCA. Groundwater depletion, for instance, may be an assessment endpoint associated with a quantity of groundwater used to manufacture of paper products.
Classification	The process whereby inventory data are assigned to impact categories (e.g., photochemical smog, lung disease, fossil-fuel depletion) under primary impact groups (e.g., ecosystem, human health, and natural resources). For example, CO_2 emissions may be classified into the greenhouse effect category under primary ecosystem impacts group.
Conversion Models	Models that help to characterize environmental impacts based on the data obtained from an inventory analysis. An example of a conversion model is the Mackay Unit World Model, which uses a generic computer fate-and-exposure model to characterize the partitioning and transformations of chemical substances introduced into a hypothetical 1 km^3 "ecosystem box."
Characterization	The assessment and possible estimation of the magnitude of environmental impact. Characterization involves the use of specific impact assessment tools, known as conversion models and impact descriptors.
Direct Impact	A potential impact that is directly attributable to an inventory item. A direct impact associated with ozone emissions could be photochemical smog.
Goal Definition and Scoping	A discrete activity in the LCA process, which may be reevaluated or modified at any point, that involves defining the study purpose and objectives; identifying the product, process, or activity of interest; identifying the intended end-use of study results; and key assumptions employed.
Impact	A potential ecosystem, human health, or natural resource effect associated with an inventory item. Acid deposition, for instance, may be an impact to the natural environment associated with X tons of SO_2 emissions identified in the inventory analysis.
Impact Assessment	A quantitative and/or qualitative process to classify, characterize, and value impacts to ecosystems, human health, and natural resources based on the results of an inventory analysis.

Impact Network	The conceptual, qualitative linking of inventory items to potential direct and indirect impacts. For instance, NO_x emissions listed in the inventory may be linked to acid precipitation, which in turn may be linked to tree damage, acidification of lakes, soil leaching, and corrosion of materials.
Impact Descriptor	A measure or set of significant environmental attributes associated with a particular impact or impact category. For example, a CO_2 emissions value from an inventory could be run through the appropriate conversion model to yield the potential level of greenhouse gas build up or global warming.
Improvement Assessment	A process to identify and evaluate opportunities for achieving improvements in products and/or processes that result in reduced environmental effects, based on the results of an inventory analysis or impact assessment.
Indirect Impact	A potential impact that is not directly attributable to an inventory item, but rather stems from another impact. Human respiratory damage, for instance, could be indirect impacts of photochemical smog, which is a direct impact of ozone emissions.
Input	A raw material, energy, or other resource requirement of a product system. Inputs may include the amount of timber required to produce 1 ton of paper, the amount of natural gas required per unit of plastic production, or the amount of soil erosion per activity.
Inventory Analysis	A process of identifying and quantifying, to the extent possible, resource and energy inputs, air emissions, waterborne effluents, solid waste, and other inputs and outputs throughout the life cycle of a product system. The inventory may include such items as the tons of CO_2 released per unit of production, the amount of coal per unit of production.
Life-Cycle Assessment (LCA)	A holistic approach to evaluating the environmental burdens associated with a product system by identifying inputs from and outputs to the environment; assessing the potential impacts of those inputs and outputs on the ecosystem, human health, and natural resources; and identifying and evaluating opportunities for achieving improvements. LCA consists of four complementary components—goal definition and scoping, inventory analysis, impact assessment, and improvement assessment.

Measurement Endpoint	A measurable response to an environmental loading that may act as a surrogate measure, quantitative or qualitative, for a related assessment endpoint. For example, acid precipitation could be a possible measurement endpoint for the assessment endpoint of lost recreation revenue at lake X that is indirectly attributable to NO_x emissions. A different measurement endpoint for this scenario could be the lost recreation revenue at lake Y due to NO_x emissions.
Nonthreshold Assumption	The concept that although recognizing any single inventory item within a given product system as a significant contributor to specific impacts is difficult, that inventory item nonetheless contributes to impacts when placed in the context of other product systems, and may therefore need to be considered in impact assessment.
Output	Air emissions, waterborne effluents, solid waste, or other releases to the environment associated with the life cycle of a given product system. Outputs can include the quantity of CO_2 released per unit of production, the volume of solid waste per unit of time, and the level of noise or odor associated with a particular activity.
Production System	An operation or group of operations associated with the production of a product or service that has clearly delineated input and output boundaries and includes operations associated with each life-cycle stage. The product system associated with polyethylene production, for instance, includes not only the company manufacturing the polyethylene, but all of the intermediate companies that produce the materials for the polyethylene production, such as the oil refinery and a natural gas transportation company.
Valuation	The explicit and collective process of assigning relative values and/or weights to potential impacts of concern (assessment endpoints). Analytic methods, for example, such as the Analytic Hierarchy Process (AHP) may be used to estimate the relative importance (value) of various impacts or impact categories to multiattribute decisions.

Appendix D

Bibliography

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16. ABSTRACT

Life-Cycle Assessment (LCA) is a holistic concept and approach for evaluating the environmental and human health impacts associated with a product, process, or activity. A complete LCA looks upstream and downstream, identifies inputs and outputs, and assesses the potential effects of those inputs and outputs on ecosystems, human health, and natural resources.

This report presents a conceptual framework for conducting a life-cycle impact assessment (LCIA), discusses major issues, and summarizes existing methods. It also identifies some of the advantages, and disadvantages of various methods.

17. KEY WORDS AND DOCUMENT ANALYSIS

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